QUANTITATIVE IMPACTS OF LAKE-LEVEL STABILIZATION ON SEDIMENT AND NUTRIENT DYNAMICS: COUPLING LIMNOLOGY WITH MODELING

By

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TABLE OF CONTENTS

Abstract of Dissertation Presented to the Graduate School of the University of Florida in Partial Fulfillment of the Requirements for the Degree of Doctor of Philosophy

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Spillways at lake outlets are used to reduce water-level fluctuations and promote year-round lake access. Such hydraulic alteration may cause accelerated sedimentation in the lake. The impact of a 1967 spillway was quantified in eutrophic Newnan's Lake (Florida) using a model and experimental work.

A transect of sedimentary profiles, dated with ^{210}Pb and ^{137}Cs by γ-ray spectroscopy, showed threefold increases in accumulation rates of organic matter, total Kjeldahl nitrogen (TKN), and total phosphorus (TP) 1200 m lakeward of the spillway since its construction. Concentrations of TKN and TP increased 3.5 and 2.4 times, respectively, in sediments deposited since 1967. These increases were

progressively less at stations farther from the spillway. Postspillway accumulation of TP was focused toward the dam, while recent TKN deposition was similar lakewide.

A 90-day spillway removal flushed 60 tons (dry weight) of sediment from the lake. Discharge concentrations of particulate organic matter, TKN and TP were highest during the first month of the drawdown under adequate hydraulic head. Storms stirred the water column and promoted flushing of resuspended matter. Field and laboratory tests did not show net oxidative removal of organic matter from exposed lake bottom. Consolidated sediments remained moderately firm after reflooding. Lowering the water depth from 62 to 30 cm in the littoral zone removed 6.08 g $m^{-2}day^{-1}$ of organic littoral sediment, likely due to increased wind-wave action on this substrate. Littoral sediments with low bulk density eroded fastest and bulk density of remaining substrate increased by an average of 250%. Redistribution of this material to deeper portions of the lake was not demonstrated.

Results from experiments, literature and theory were synthesized using feedback dynamics. The resulting simulation model was most sensitive to changes in sediment resuspension and actions of bacteria and benthic heterotrophs on phosphorus dynamics in surface sediments. Research aimed at quantifying these processes will enhance confidence in the model. Separate impacts of the spillway (decreased flushing, littoral zone vegetation, and sediment resuspension) had minor implications for model behavior. Combined, however, they increased seston, water-column phosphorus, and sedimentation by 19, 39, and 27%, respectively.

In conclusion, after 25 years, intended benefits of the spillway (lake access, navigation) have been replaced with lake management "costs" such as increased turbidity, primary production, and sedimentation.

CHAPTER 1
INTRODUCTION

Effects of spillways

Spillways have been built at the outlets of many large shallow lakes in Florida during the past 50 years (Bishop 1967; Davis 1973). The majority of these water control structures were designed to prevent low lake-stages during the dry winter season characteristic of Florida's climate. Year-round access to these lakes was promoted, and their use for irrigation, recreation, and sportfishing maximized. The structures varied from simple earthen dams to stoplog weirs and concrete spillways with or without removable boards.

The hydraulic modification produced by these dams may produce short-term benefits for lake access and navigation, but long-term consequences for lake ecosystems are poorly understood. Stabilized lake levels may accelerate the accumulation of sediment and nutrient-rich detritus in the lake. This can occur in several ways.

First, the design of most spillways eliminates discharge of bottom-water. This layer of water is rich in solids due to sediment resuspended by wind-wave induced currents (Sheng and Lick 1979; Paul et al. 1982). Outflow is restricted to less turbid surface water flowing over the crest of the spillway. During extended periods of low rainfall when outflow of the lake is confined to evapo-transpiration and losses to groundwater, suspended solids are not exported.

Second, decreased water-level fluctuations reduce exposure of littoral sediments to air, thus preventing seasonal consolidation of this flocculent substrate and reducing oxidative removal of organic matter. These effects may be particularly pronounced in shallow lakes with vast littoral areas. Such consolidation has been documented when the water level in a lake was artificially lowered for several months. Organic and nutrient-rich sediments from Lake Apopka (Florida), with an initial water content in excess of 80%, consolidated 40-50% after exposure to rain and sun for 5 months (Fox et al. 1977). Significant consolidation of the top 5-10 cm of exposed littoral substrate was noted during the drawdown of Fox Lake (McKinney and Coleman 1980). Oxidation of exposed lake sediments has been suggested (Harris and Marshall 1963; Wegener and Williams 1974), but experimental evidence is controversial. No significant oxidation was noted during drying of Lake Apopka sediments (Fox et al. 1977), while Holcomb et al. (1975) reported a considerable reduction in organic matter content of exposed sediments in Lake Tohopekaliga (Florida).

Third, stabilized lake levels may reduce wetland habitat around the lake (Keddy and Reznicek 1985). Many wetland plants require seasonal fluctuations in water level for growth and reproduction (Gosselink and Turner 1978; Bedinger 1979). Constant water levels result in a gradual decline of such wetland taxa and may alter the wetland's ability to filter nutrients and sediments carried into the lake in runoff from the watershed.

Increased accumulation of flocculent sediment and nutrient-rich detritus may significantly affect lake ecosystems. Unconsolidated sediments are easily resuspended (Lam and Jacquet 1976; Sheng and Lick 1979) resulting in increased turbidity in the water column and reduced light penetration. This may change plant communities and reduce submerged vegetation. Periodic resuspension of sediment and associated pore-fluid also may lead to increased availability of nutrients for algal utilization in the trophogenic layer (Holdren and Armstrong 1980; Pollman 1983). Organic and inorganic suspended matter exert considerable biological and chemical oxygen demand (Hargrave 1969; James 1974) which stresses heterotrophic communities in the lake and promotes nutrient release from the sediments (Mortimer 1971; Theis and McCabe 1978; Stauffer 1981). Deposits of flocculent sediment in the littoral zone eliminate firm substrate for rooted macrophytes, thereby reducing refuge and feeding habitat for fish. Such accumulations may also reduce the availability of preferred nesting habitat for centrarchids, such as the largemouth bass (<u>Micropterus salmoides</u>) and black crappie (<u>Pomoxis nigromaculatus</u>) (Bruno 1984), which rely on firm substrate for deposition of eggs (Eddy and Underhill 1978).

Related Studies

In spite of these plausible effects, little is known about the long-term response of a lake to damming of its outlet. Much work has focused on short-term responses following drastic water-level manipulations such as the filling of a reservoir by damming of a stream or the drawdown of an existing impoundment. Many of these investigations are reviewed in Baxter (1977) and Cooke et al. (1986). Those applicable to this work are referred to in Chapters 3 and 4.

Long-term investigations have concentrated on changes in littoral flora following water-level stabilization and the possible effects of such a reduction in high water-low water perturbation on the ontogeny of aquatic ecosystems. Symoens et al. (1988) reported a reduction in species diversity in the littoral zone of eutrophic Lake Virelles (Belgium) due to water-level stabilization by an outflow control structure. Plants characteristic of the periodically exposed shores disappeared. These included sedges, rushes, and members of the buckwheat family. Fluctuating water-levels increased the diversity of vegetation types and plant species along the shoreline of the Great Lakes (Keddy and Reznicek 1985). Low water periods allowed many mud-flat annuals and emergent marsh species to regenerate from buried seeds. Water-level stabilization in Lake Tohopekaliga (Florida) eliminated a considerable portion of the vegetated flood plain (Holcomb and Wegener 1971).

Effects of stabilizing water-levels on the ontogeny of aquatic ecosystems have been described for wetlands (Gottgens and Montague 1988; Gunderson 1989; Walters et al. 1992), aquaculture systems (Odum 1971), and floodplain vegetation (Goodrick and Milleson 1974). Freshwater marshes, such as the Florida Everglades, appear to be maintained at an early successional stage by seasonal fluctuations in water levels and other pulsing factors (e.g. fire, grazing). Aerobic decomposition of accumulated organic matter may be promoted when sediments are exposed during the dry season. Nutrients, released upon reflooding, support a wet-season bloom in productivity. A fluctuating water regime may be a dominant force in preventing organic matter build-up and maintaining an aquatic ecosystem in a state of pulsed stability (Odum 1971).

Research Methods: Coupling Limnology with Modeling

Two research approaches are convenient in an investigation of long-term changes in a lake following spillway installation. Both paleolimnology and ecosystem modeling work with flexible time-frames and allow investigation of long-term and slow-acting processes. Paleolimnology is particularly useful when a detailed historical data base is absent.

Paleolimnologists assume that lake sediments accumulate in an orderly fashion through time. These sediments contain a relatively stable, historical record of past conditions of the lake-watershed ecosystem. This record may be in the form of physical, chemical, and/or biological signals. The timing of these signals requires dating of levels in sediment cores which can be accomplished, for instance, by measuring the content of radioactive carbon (^{14}C), lead-210 (^{210}Pb), or cesium-137 (^{137}Cs). The age of recent sediments (up to 120 years) can be estimated by the content of short-lived isotopes (e.g. ^{210}Pb) (Eakins and Morrison 1978; Krishnaswami and Lal 1978). In this work, paleolimnological techniques are applied to quantify the impact of a 1967 spillway on sediment composition and net rates of sediment and nutrient accumulation in Newnan's Lake (Chapter 2).

Knowledge of past lake conditions based on interpretations of the sedimentary record is not only useful to evaluate rates of change in the lake (such as loss of volume), but it is also helpful in the development of lake management models, because it can assist in reconstructing the reasons for past changes in the lake. Models of different hypotheses of the causes of past lake responses (such as an increase in the rate of sedimentation) to perturbation (such as the construction of a dam) may be used to predict the effect of future management actions. One of the objectives of the simulation model presented here (Chapter 5) is to evaluate the long-term effect of spillway construction in the outlet of Newnan's Lake on variables such as sedimentation rate, lake depth, and concentrations of nutrients and suspended matter in the water column. In addition, the model can provide options for lake management by testing the effects of alternative management strategies (such as a modified dam or a drawdown) on intended uses of the lake.

The scheduling of a short-term, experimental drawdown in Newnan's Lake in the spring of 1989 provided an opportunity to investigate the impact of this management strategy. This investigation focused on the effect of the drawdown on littoral and profundal lake sediments for two reasons. First, the accumulation of flocculent and nutrient-rich sediments in the lake may have significant implications for the lake ecosystem (as outlined earlier). This material may be removed during a drawdown by flushing of suspended matter and/or oxidation of exposed littoral substrate. Both these processes were quantified in this work (Chapter 3). Second, by focusing on the sediments, the effectiveness of the drawdown to accomplish its main objective may be determined. The main objective was to improve littoral habitat for fish growth and recruitment (Krummrich, Florida Game and Fresh Water Fish Commission, pers. comm.). This improvement may occur when consolidation, and/or removal of flocculent littoral substrate promotes the establishment of littoral plant communities which may, in turn, provide refuge and feeding habitat for fish. Firm littoral substrate also provides preferred nesting habitat for species of the sunfish family (Centrarchidae) (Bruno 1984), economically important sportfish in Newnan's Lake. This study measured both consolidation of exposed littoral substrate, and erosion and subsequent "sloughing" of this material to deeper parts of the lake (Chapter 4).

Finally, studying processes in the lake with and without the spillway complements the modeling exercise. It assists in determining coefficient values for modeling the effect of the spillway on the lake, and it allows field testing of the model by comparing the results of the drawdown study with the results of the same change made in the model. As such, the limnological work assists in model development and analysis. By forcing the quantification of processes in the lake, the modeling exercise may reveal gaps that exist in our understanding of this ecosystem and identify necessary field or laboratory investigations.

Study Area

Newnan's Lake is located 8 km east of Gainesville, Florida, within the Orange Creek Basin (Figure 1-1). The geomorphology of this basin was first described by Pirkle and Brooks (1959). It is dominated by the Hawthorne formation, a marine deposit of Miocene age consisting of phosphatic sands, clays, and limestone. It is relatively impermeable compared with the underlying limestone and acts as a confining layer. Surface flow, subsurface flow through sands and clayey-sands, and direct rainfall are the major sources of water for the lake (Canfield 1981). The drainage area north of the lake supplies surface water inflow via Hatchet Creek, Little Hatchet Creek, and several smaller streams. The low mineral content of the lake water suggests a small input from groundwater. Loss of water from the lake is dominated by evapo-transpiration in Florida's warm climate. Of the 130 cm of rainfall received by the Orange Creek watershed annually (Adkins 1991), only 10% leaves as surface drainage (Clark et al. 1964).

Land-use in the basin is dominated by forest (Table 1-1), including large tracts of commercial pine (_Pinus elliotti_) plantations. Extensive areas of undeveloped pine flatwoods also occur, with riverine hardwood-swamp and

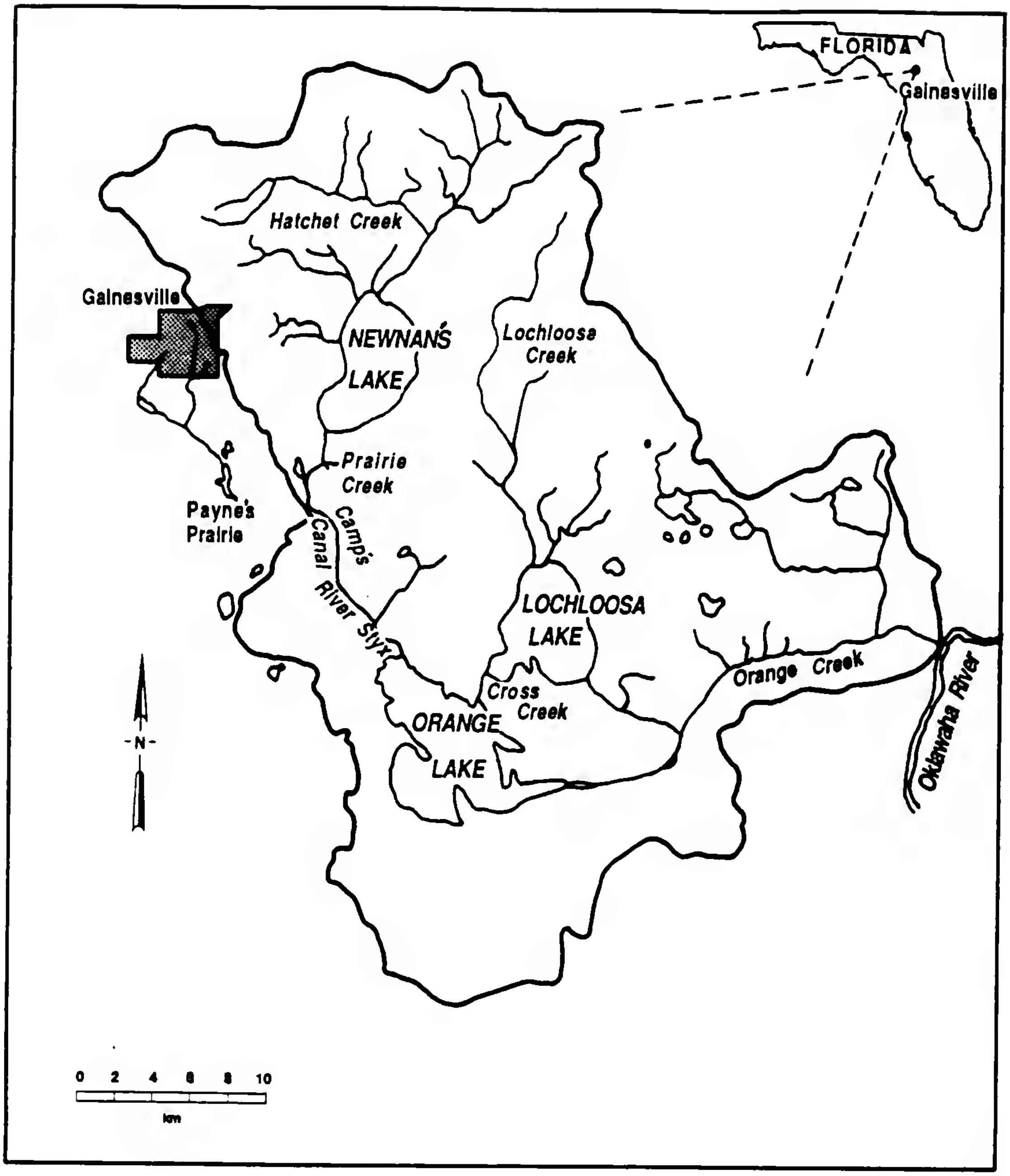


Figure 1-1. Drainage map of the Orange Creek Basin.

cypress (Taxodium distichum) along the water courses. A prominent fringe of cypress trees borders the lake.

Table 1-1. Land-use data for the Newnan's Lake watershed. Values in percent of total watershed area.

Urban	8.6
Forest [1]	75.6
Agriculture	7.7
Open water	8.0
Wetland	0.1

Source: Huber et al. 1982.
[1] Includes areas dominated by cypress trees.

Newnan's Lake has a surface area of approximately 3000 ha and a maximum and mean depth of 3.6 m and 1.5 m, respectively (Nordlie 1976) (Table 1-2). It is a flat bottom basin with a few depressions where the water depth exceeds 2.5 m at average lake stage (Figure 1-2). The lake has a single major surface water outlet, Prairie Creek, which drains to the south (Figure 1-1) through an area with extensive marsh and bottomland hardwood communities. Prior to the construction of Camp's Canal (in the early 1930s) Prairie Creek discharged into Payne's Prairie (Figure 1-1). This bypass canal was constructed around the eastern margin of the prairie to control flooding and manage the land for cattle grazing (Gottgens and Montague 1988). The canal

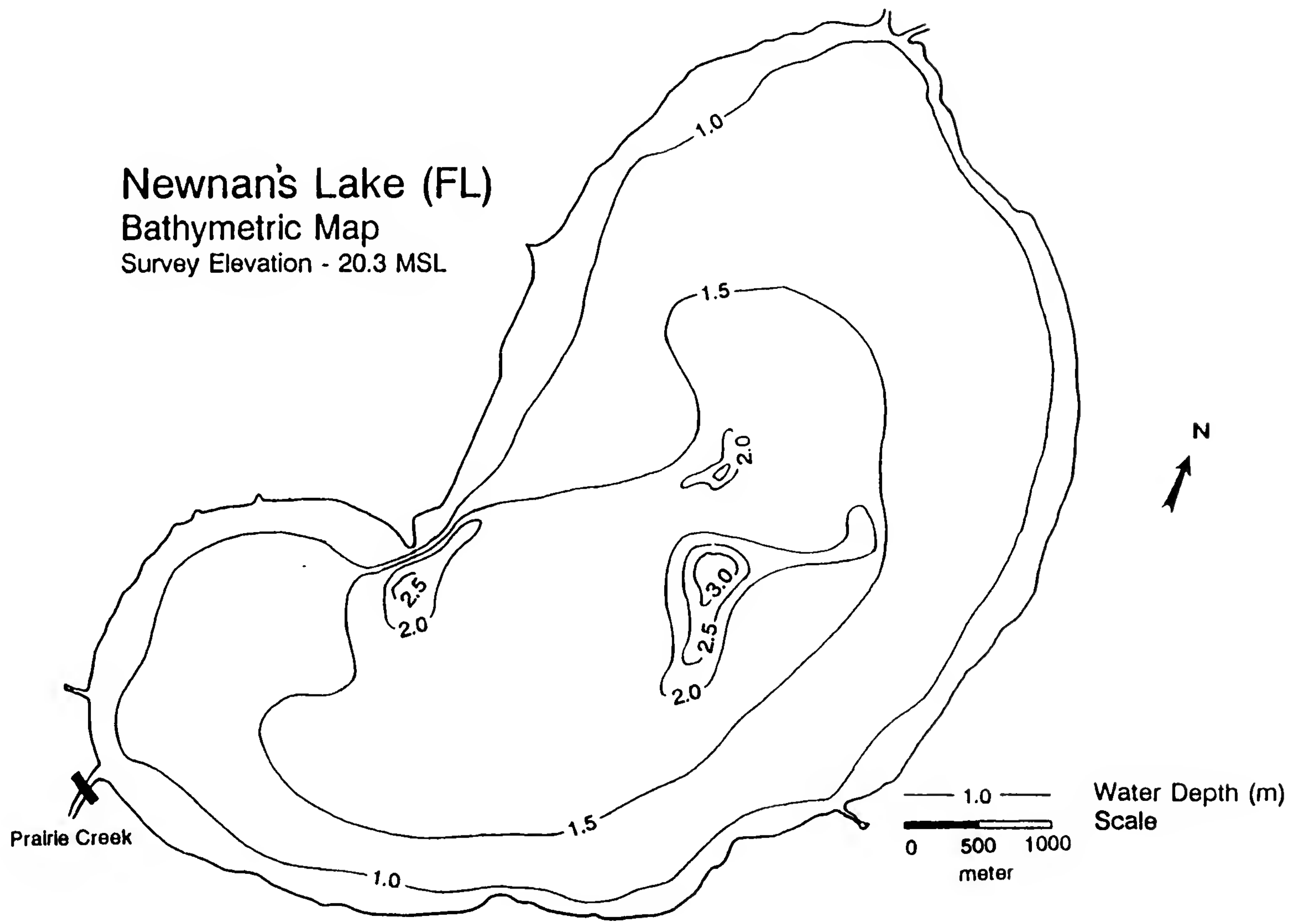


Figure 1-2. Bathymetric map of Newnan's Lake (drawn from transect data from Skoglund, 1990).

directs the water around the prairie on a sluggish course to the River Styx and subsequently to Orange Lake (Figure 1-1). The prairie has been managed as a state park and marsh preserve since 1970 and a limited amount of water from Prairie Creek has been diverted back into the preserve.

Table 1-2. Morphometry of Newnan's Lake.

Surface Area	3042	ha
Maximum Depth	3.6	m
Mean Depth	1.5	m
Development of Shoreline	1.09	
Drainage Basin Area	308	km^2
Lake Volume	58×10^6	m^3
Lake Detention Time	0.6	yrs

The outflow from Newnan's Lake through Prairie Creek has been regulated with a spillway since 1967. This 50 m long structure consists of removable flashboards down to the channel bed. The elevation of the top of the spillway boards prevents surface discharge when the water level in the lake drops below 20.1 m (MSL). During this study, one of the top boards was missing, lowering the elevation of a 1.2 m section of the top of the spillway boards to 19.9 m (MSL).

A comparison of pre- and postspillway stage-duration curves indicates a reduction in the range of water-level fluctuations of approximately 32 cm or 30% (Skoglund 1990;

Adkins 1991). This reduction is more pronounced if severe prespillway droughts are included, during which no water level records were kept (1954-56 and 1962-64). In this shallow lake, with a very small elevation gradient in the littoral zone, such reduced water-level fluctuations may significantly affect seasonal exposure of littoral sediments and the area of vegetated littoral habitat. The spillway has raised the average stage of the lake by 13 cm, i.e. it has increased the average water depth in the lake by 9%.

Newnan's Lake may be classified as an eutrophic, soft water lake (Canfield 1981). Water quality data show high nutrient concentrations, high and variable chlorophyll a values, and low Secchi disk transparency (Table 1-3). High color (often in excess of 150 Pt units) occurs as a result of input of water rich in dissolved humic materials from the surrounding flatwood and cypress communities.

Temperature and dissolved oxygen data for Newnan's Lake were also typical of shallow, productive systems. Vertical changes in water-column temperature rarely exceeded 3°C, regardless of season (Crisman 1986a). Consequently, thermal stratification rarely, if ever, occurred. Daytime surface water during summer and early fall was often supersaturated, while the water column in the deeper portions of the lake below 3 m approached anoxia

Table 1-3. Summary of water quality data for Newnan's Lake. Numbers in parentheses are the minimum and maximum values measured.

Parameter	Unit	Source			
		Shannon (1970) 1)	Canfield (1981) 2)	Crisman (1986a) 3)	During this study 4)
pH			6.8 (6.7–6.9)	7.6 (6.3–8.9)	7.9 (6.9–8.9)
Tot. Alkalinity	mg/l CaCO3		14 (12–18)		18 (17–19)
Conductivity	umhos/cm	60 (45–73)	59 (54–64)		91 (79–100)
Total N	mg/l	1.41 (0.66–2.35)	1.30 (0.88–1.50)	2.1 (0.5–3.2)	1.93 (1.43–2.64)
NH3–N	mg/l	0.36 (0.03–0.80)			
NO3–N	mg/l	0.07 (0.02–0.22)			
Total P	mg/l	0.11 (0.08–0.15)	0.05 (0.02–0.08)	0.17 (0.02–0.40)	0.07 (0.06–0.09)
PO4–P	mg/l			0.01 (0.0–0.22)	0.03 (0.01–0.05)
Chlorophyll a	mg/m3	47.4 (26.1–86.8)	38.0 (24.1–55.0)	52 (11–157)	25.1 (1.1–56.7)
Color	Pt units	189 (61–245)	93 (45–150)		94 (40–200)
Secchi	m	0.5 (0.4–0.6)	0.6 (0.5–0.7)	0.55 (0.15–0.95)	0.60 (0.31–0.81)

1) Sampling period June 1969–June 1970, every two months. Three stations were sampled and composited per collection date.
2) Based on collections at three dates from September 1979–August 1980. At each date, three mid-lake samples were taken at a depth of 0.5 m. Each sample was analyzed separately.
3) Based on collections from January 1979–January 1983. Sampling frequency varied between 3-12 per year. Three different samples were analyzed and averaged.
4) Based on collections during 1989. Data retrieved from STORET, a data base of sampling sites and their associated water quality data. Sampling depth was 0.5 m.

as a result of intense decomposition in surficial sediments (Crisman 1986a).

Using the data sources from Table 1-3, the ratio of water-column N:P (in grams) varied from 12.4 to 27.6. This ratio is approximately 7:1 in aquatic plant material (Vallentyne 1970). Therefore, these water quality data indicate that phosphorus is generally less available than nitrogen relative to their content in aquatic plant material, and that primary production in Newnan's Lake is likely limited by phosphorus.

The lake had dense growths of filamentous algae dominated by blue-greens (Cyanophyta) such as *Spirulina* sp., *Anabaena* sp., and *Aphanizomenon* sp.(Nordlie 1976; Crisman 1986a). At times, macrophytes dominated by exotics such as water hyacinth (*Eichhornia crassipes*) and hydrilla (*Hydrilla verticillata*) were abundant. Since the early 1970s, periodic applications of herbicides have been made to restrict the spread of these exotics. The lake bottom was covered with a homogeneous layer of highly flocculent, organic sediment (Holly 1976). A review of existing literature and an inventory of the existing data base for Newnan's Lake is given in Gottgens and Montague (1987a), and in its accompanying categorized bibliography (Gottgens and Montague 1987b).

CHAPTER 2
QUANTITATIVE IMPACTS OF LAKE LEVEL STABILIZATION ON MATERIAL TRANSFER BETWEEN WATER AND SEDIMENT

In Chapter 1, it was hypothesized that spillways accelerate the accumulation of sediment and nutrient-rich detritus in a lake and that such increased trapping of material may have considerable long-term impact on the lake ecosystem. Little is known, however, about increases in the rate of sedimentation caused by damming of a lake outlet. Measurement of this rate is essential for determining loss in lake volume and quantifying mineral cycling in lakes, including the modeling of nutrient concentrations in water (Dillon and Rigler 1974: Vollenweider 1976). It is also needed for the development of a sediment or pollutant budget.

Material accumulation rates can be calculated by determining geochronology in the sediment profile using radioactive decay of fallout ("unsupported") lead-210 (^{210}Pb) following burial in sediments (Appleby and Oldfield 1978). Matching the occurrence of cesium-137 (^{137}Cs) in the profile with the onset of widespread atmospheric nuclear testing in the early 1950s (Ritchie et al. 1973) provides an additional age-marker. These paleolimnological techniques have been used widely to detect changes in sediment accumulation rates caused by urban development in a watershed (Smeltzer and Swain 1985), major storm events (Robbins et al. 1978), and deforestation (Oldfield et al. 1980). They are particularly helpful in gaining insight into past conditions in lakes when a long-term data base is absent. Here, they were applied to quantify the impact of a 1967 spillway on sediment composition and net accumulation rates of organic matter, nitrogen, and phosphorus in Newnan's Lake.

Materials and Methods

Collection of Sediment Cores

Core locations were arranged in a transect of increasing distance lakeward from the spillway (Figure 2-1). Cores 8, 6, and 1 were taken, respectively, 1200, 3000, and 4500 m lakeward of the structure at a water depths of 115, 150, and 150 cm. The cores were collected from the lake in the spring of 1989 using a Livingstone piston corer (Livingstone 1955) equipped with 4.1 cm diameter cellulose butyrate tubes. The piston was positioned at the lower end of the coring tube which was then lowered to the level at which sampling was to commence. During this operation the piston cable was payed out freely, but once the sampling level was reached the cable was secured to the boat to prevent the piston from moving any farther down. The tube was driven into the

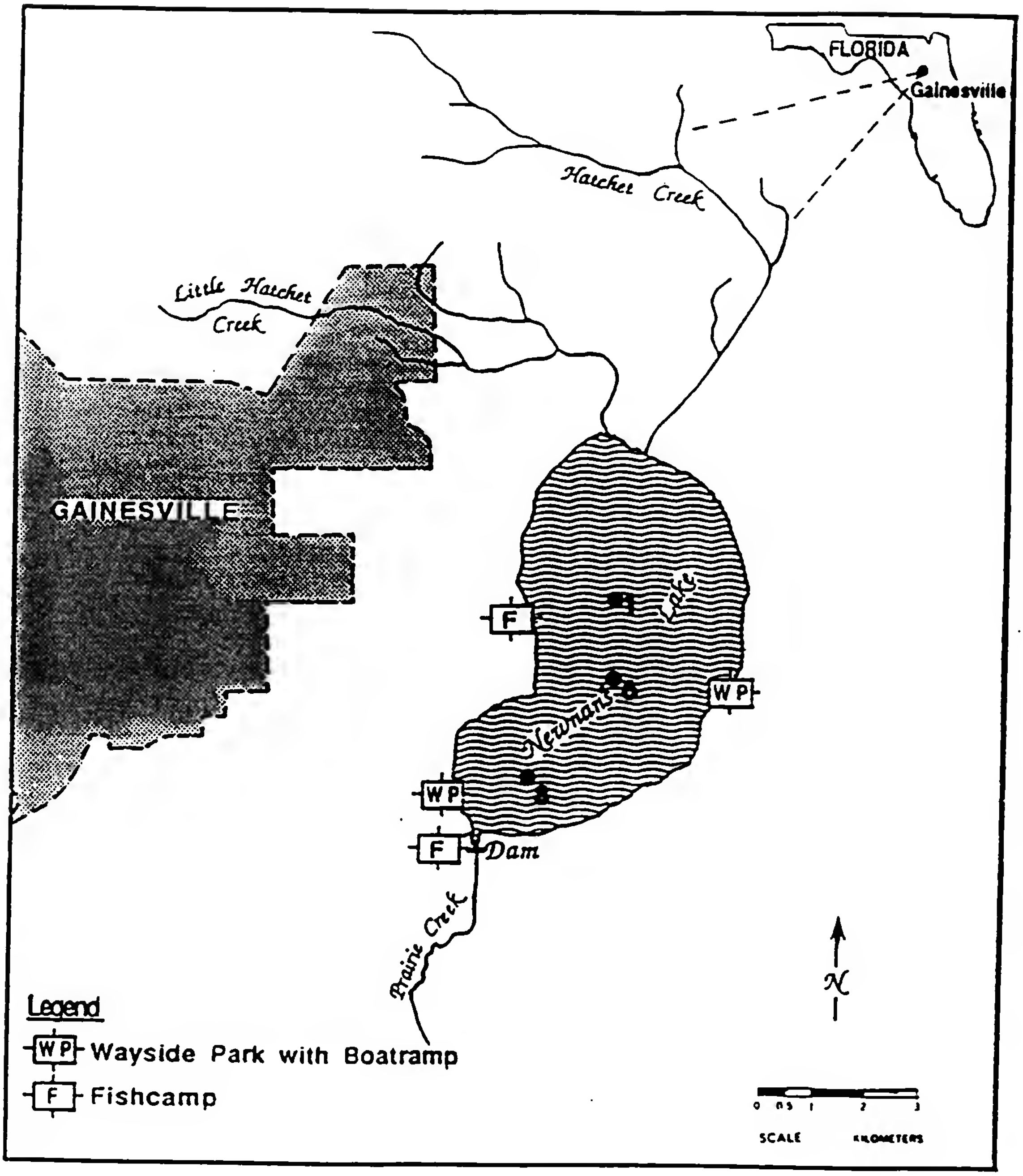


Figure 2-1. Insert map of Florida showing location of study area and map of Newnan's Lake with core sites indicated (filled circles).

sediments by pushing on the extension rod while the piston prevented the sample from being compressed or lost upon retrieval. After the apparatus was hoisted to the surface, appropriate caution was exercised to preserve the sediment-water interface. Cores were kept refrigerated until they were sectioned. The cores were sectioned in 1 cm intervals, which were stored in plastic bags at 4°C.

Radio-isotope Analyses

Concentrations of ^{210}Pb and ^{137}Cs were measured by direct γ-assay using a coaxial N-type, intrinsic-germanium detector (Princeton Gamma Tech). The counting system used for spectral analysis (Figure 2-2) is located in the University of Florida's Department of Environmental Engineering Sciences' Low Background Counting Room. This room was designed to reduce the level of background radiation interference during sample counting. An outer shield (0.95 cm steel), main shield (10.1 cm lead), and an inner lining (0.05 cm cadmium + 0.15 cm copper) were used to reduce background radiation at the germanium detector. The detector has a 38% efficiency (at 1332.5 keV) and operates at a temperature near that of liquid nitrogen (-196°C), which is provided by a high-vacuum cryostat dewar system (capacity 30 l). This thin-window (beryllium), standard detector has a relatively low efficiency resulting in the need for long count times. This type of detector, however, counts over a large range of γ-energies, is

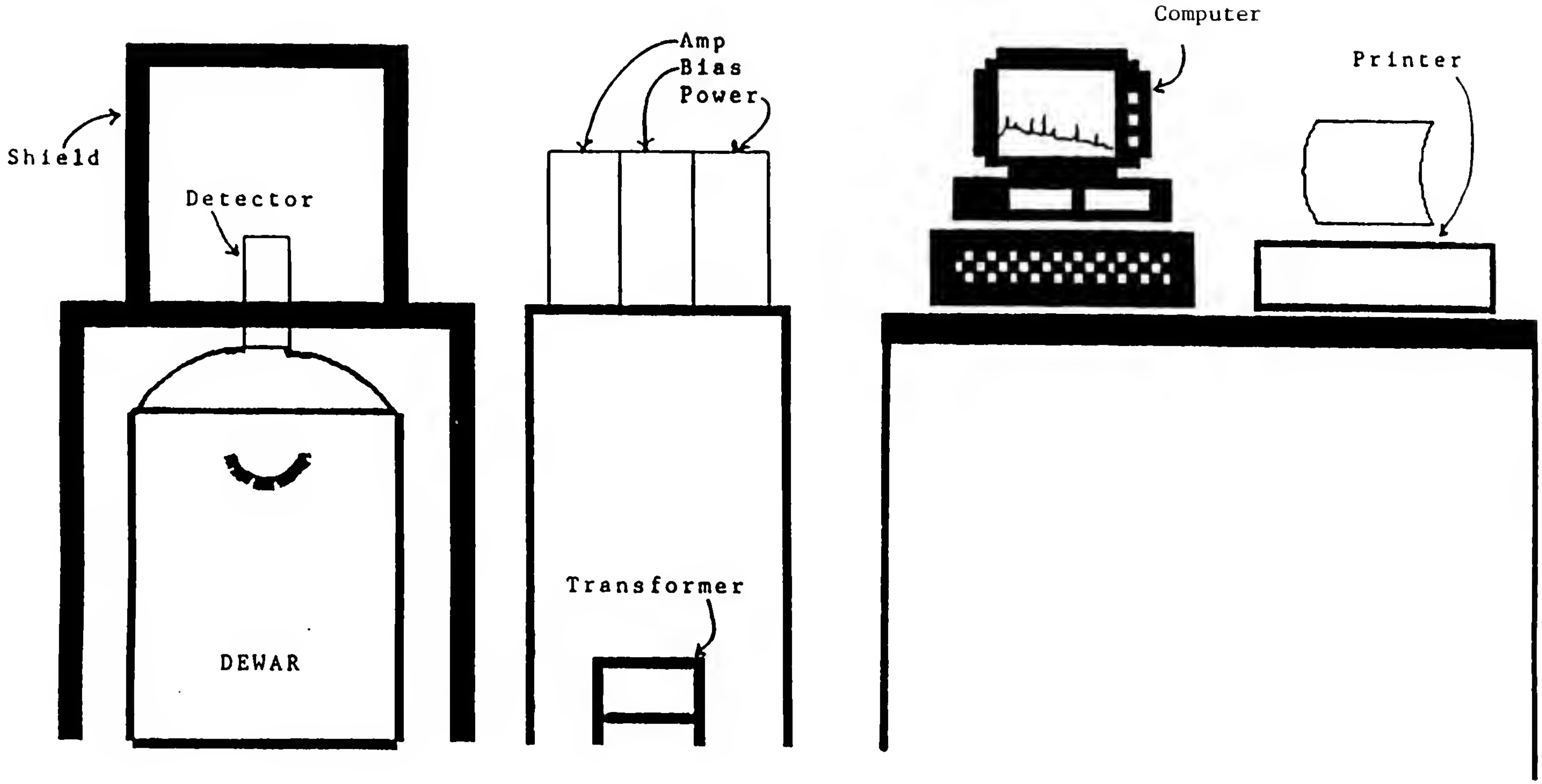


Figure 2-2. Diagram of the counting system at the Department of Environmental Engineering Sciences, University of Florida. (After Nagy 1988)

suitable for many different sample configurations, and therefore more generally useful. It also detects efficiently other gammas from the ^{238}U decay series and can thus be used to determine supported and unsupported levels of ^{210}Pb simultaneously (Appleby et al. 1986; Nagy 1988). The electronics include a preamplifier (RG11B/C, Princeton Gamma Tech, Inc.), amplifier (TC 242, Tennelec), bias supply (5 kV, TC 950, Tennelec), power supply (TC 909, Tennelec), and transformer (Sola). A Zenith (Z159) computer with multi-channel analyzer ("Maestro" ADCAM 100, EG & G Ortec) and mathematical spreadsheet (Quattro, Borland Inc.) was used for data conversion and analysis. A new procedure developed for γ-ray spectroscopy using the Department's recently acquired P-type, intrinsic germanium well detector is given in Appendix A.

Samples for isotope analysis were dried at 95°C for 24 hours, pulverized by mortar and pestle, weighed, and placed in small plastic petri-dishes (#1006 Falcon). Core sections were combined (up to 4 cm) to obtain an adequate sample weight (generally more than 1 gram). Petri dishes were sealed with plastic cement and left for 14 days to equilibrate radon (^{222}Rn) with radium (^{226}Ra). Counting times varied from 14 to 45 hours depending on sample weight; small samples needed longer counting times to minimize uncertainty. Blanks were counted for every two samples to determine background radiation. Standards were run with the same frequency to track efficiency (counts/γ) and to calculate a ^{226}Ra conversion factor (pCi/cps). Sample spectra were analyzed for activity in the 46.5 kev (^{210}Pb) and 662 kev (^{137}Cs) peaks. Activities at 295 kev (^{214}Pb), 352 kev (^{214}Pb), and 609 kev (^{214}Bi) representing uranium series peaks were used to compute supported levels of ^{210}Pb.

Calculation of ^{210}Pb dates followed the constant rate of supply model (Goldberg 1963; Appleby and Oldfield 1978) which is able to quantify changing sediment accumulation rates. In this model, the cumulative residual unsupported ^{210}Pb, A_t, beneath sediments of age t varies according to

$$A_t = A_0 e^{-kt} \tag{2-1}$$

where A_0 is the total residual unsupported ^{210}Pb (pCi/cm^2) and k is the ^{210}Pb radioactive decay constant. A_t and A_o are calculated by numerical integration of the ^{210}Pb profile. The age of sediments of depth x is then given by

$$t = \frac{1}{k} \ln \frac{A_0}{A_t} \tag{2-2}$$

The sedimentation rate (r) can be calculated directly (Appleby & Oldfield 1978):

$$r = \frac{kA_t}{C} \qquad (2\text{-}3)$$

where C is the concentration of unsupported ^{210}Pb (pCi/g) in the sediment layer of interest. This model appears applicable to Florida lakes, particularly because ^{210}Pb residuals match both the known atmospheric flux of this isotope as well as the residuals of nearby cores (Binford and Brenner 1986; Gottgens and Crisman 1992).

Bulk Density, Nutrient, and Carbon Analyses

Bulk density (mg/cm^3) and organic matter content (mg/mg) were determined on 1 cm^3 subsamples for as many as 45 intervals per core. Bulk density was measured by drying of subsamples at 95°C for 24 hours, cooling under dessication and weighing. Organic matter content was estimated by weight loss on ignition at 550°C for 1 hour followed by rehydration with distilled water and re-drying at 95°C for 24 hours (method 209D, A.P.H.A. 1985). Nitrogen was measured as total Kjeldahl nitrogen (TKN) using a Technicon-II semi-automated manifold after digestion following Bremner and Mulvaney (1982), but modified to exclude selenium as a catalyst. The digestate was also used for total phosphorus (TP) determinations. Liberated orthophosphate was determined with the ascorbic acid method (method 424F, A.P.H.A. 1985) using a Milton Roy spectrophotometer (Model 20) with a 1 cm light path.

Between 15 and 20 intervals per core were analyzed for these nutrients. Total carbon in the sediments was measured by combustion of the sample at 900°C followed by titration of the evolved CO_2 (Lee and Macalady 1989) using a model 5011 coulometer (Coulometrics, Inc.). Total organic carbon was measured using a combustion temperature of 500°C.

Results and Discussion

Results of paleolimnological analyses may be presented in units of concentration or as rates of accumulation. Concentration, expressed as a relative measure of sediment composition (e.g. mg/g dry weight), is the conventional way of expressing sediment stratigraphy (Shapiro et al. 1971; Pennington 1973; Griffiths and Edmondson 1975; Kramer et al. 1991). Such data, however, are vulnerable to variations in sedimentation of other components in the profile. For example, increased deposition of allogenic inorganic material lowers the concentration of organic matter in the sediment. Effects of such dilution can be eliminated from the calculations using ratios of components in the sedimentary matrix. Accumulation rates, on the other hand, are normalized to time thus avoiding the problem of co-variance among different sedimentary components. Detailed sediment dating with acceptable analytical precision, necessary for this procedure, can now be done with direct ^{210}Pb low-background γ-assay (Appleby et al. 1986; Gottgens and Crisman 1992). This technique eliminates uncertainties associated with ^{226}Ra supported activity and provides additional independent data markers in the profile via a simultaneous assay for other γ-emitting isotopes, including ^{137}Cs. Nonetheless, accumulation rate calculations are numerically sensitive to small errors in ^{210}Pb measurements. The uncertainty associated with this may be reduced by averaging accumulation rates over longer periods of time. In this study, both concentration and accumulation rate data are given to provide complementary information about the recent history of the lake.

Cores consisted mostly of homogeneous brown material with organic matter content generally greater than 50% and bulk density less than 70 mg/cm^3 throughout the top 30-40 cm (Gottgens and Crisman 1992). Abrupt increases in the organic matter, TKN, and TP profiles of core 8 at 33 cm depth coincided with a ^{210}Pb determined sediment age of approximately 1967, i.e. the time that the lake outflow was dammed (Figure 2-3a). Concentrations of TKN and TP increased 3.5 and 2.4 times, respectively, in sediments deposited since 1967. This signal was progressively less at stations farther from the spillway (Figures 2-3b and c). Increased nutrient concentrations at a ^{210}Pb date of approximately 1952-56 (Figure 2-3a, b, and c) matched the driest year (1954) in north Florida's recent history. This

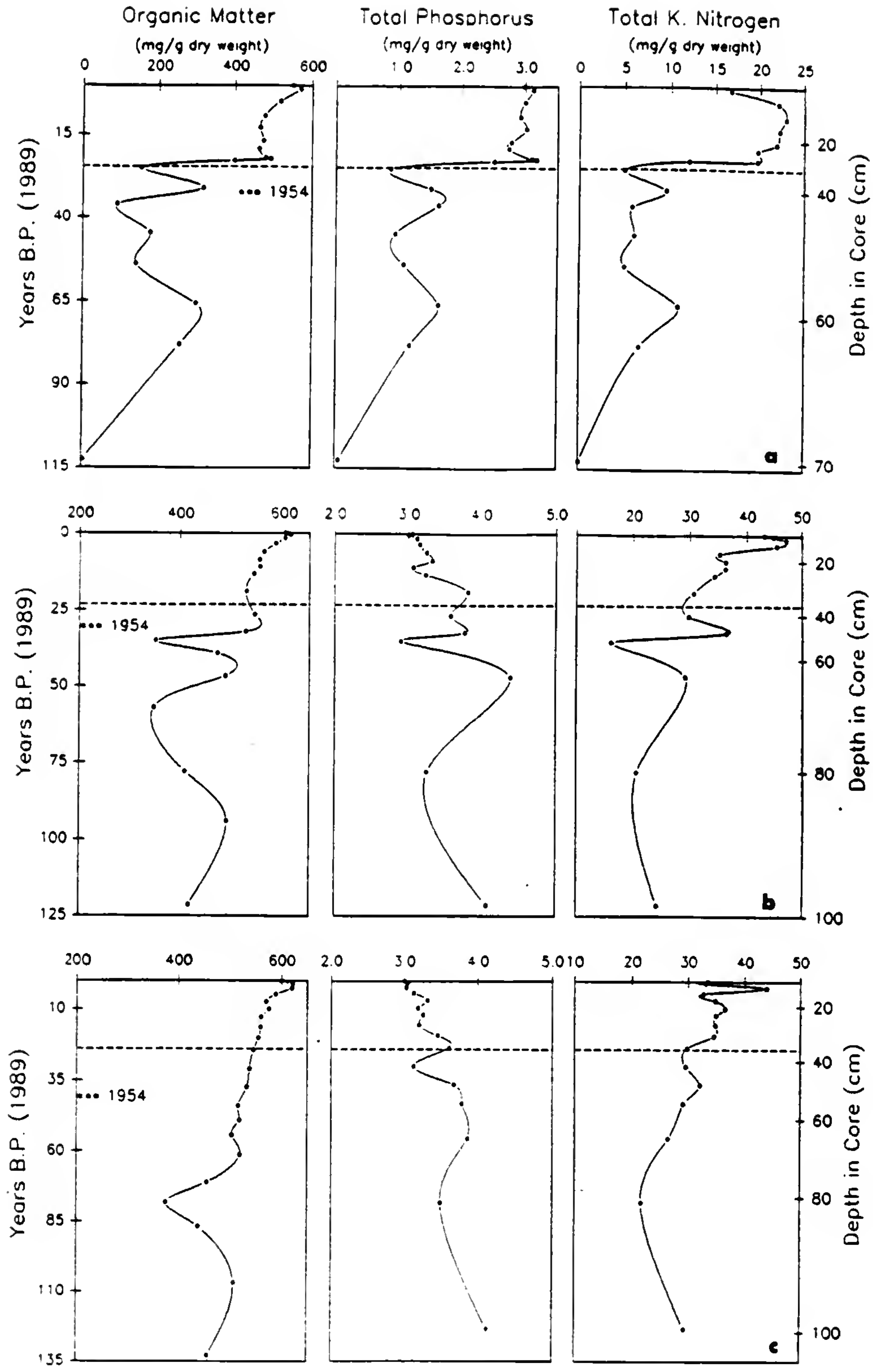


Figure 2-3. Core sites 8 (a), 6 (b), and 1 (c), Newnan's Lake: Profiles for organic matter, total phosphorus, and total Kjeldahl nitrogen. Spillway installation (---) and first occurrence of ^{137}Cs marker (***) indicated.

may have resulted from reflooding of exposed littoral sediments and subsequent release of nutrients to the water column.

The first occurrence of detectable ^{137}Cs, marking the onset of widespread atmospheric nuclear testing in the early 1950s, matched well with the determined ^{210}Pb chronology (Figures 2-3a, b, and c). This agreement is remarkable considering the shallow character of the lake and the flocculence of the bottom substrate which makes these upper sediments vulnerable to physical disturbance. Eutrophic systems such as Newnan's Lake, however, accumulate sediment rapidly, so that these periodic physical disturbances likely affect short time-intervals. The bulk density profile for core 8 (Figure 2-4) demonstrated both the fluid nature of the deposits and the sudden, fivefold decrease in bulk density after spillway installation. Bulk density in the top 33 cm of sediment (e.g. 24 years) was less than 60 mg/cm^3, and average net accumulation is 1.3 cm per year. In most sediment profiles, bulk density increases with depth due to compaction. Yet, the profile illustrates a significant transition at the time the lake outlet was dammed.

Unconsolidated sediments may be resuspended easily by wind-induced currents in shallow systems such as Newnan's Lake. The resulting organic turbidity increases sediment oxygen demand (Bowman and Delfino 1980), particularly in

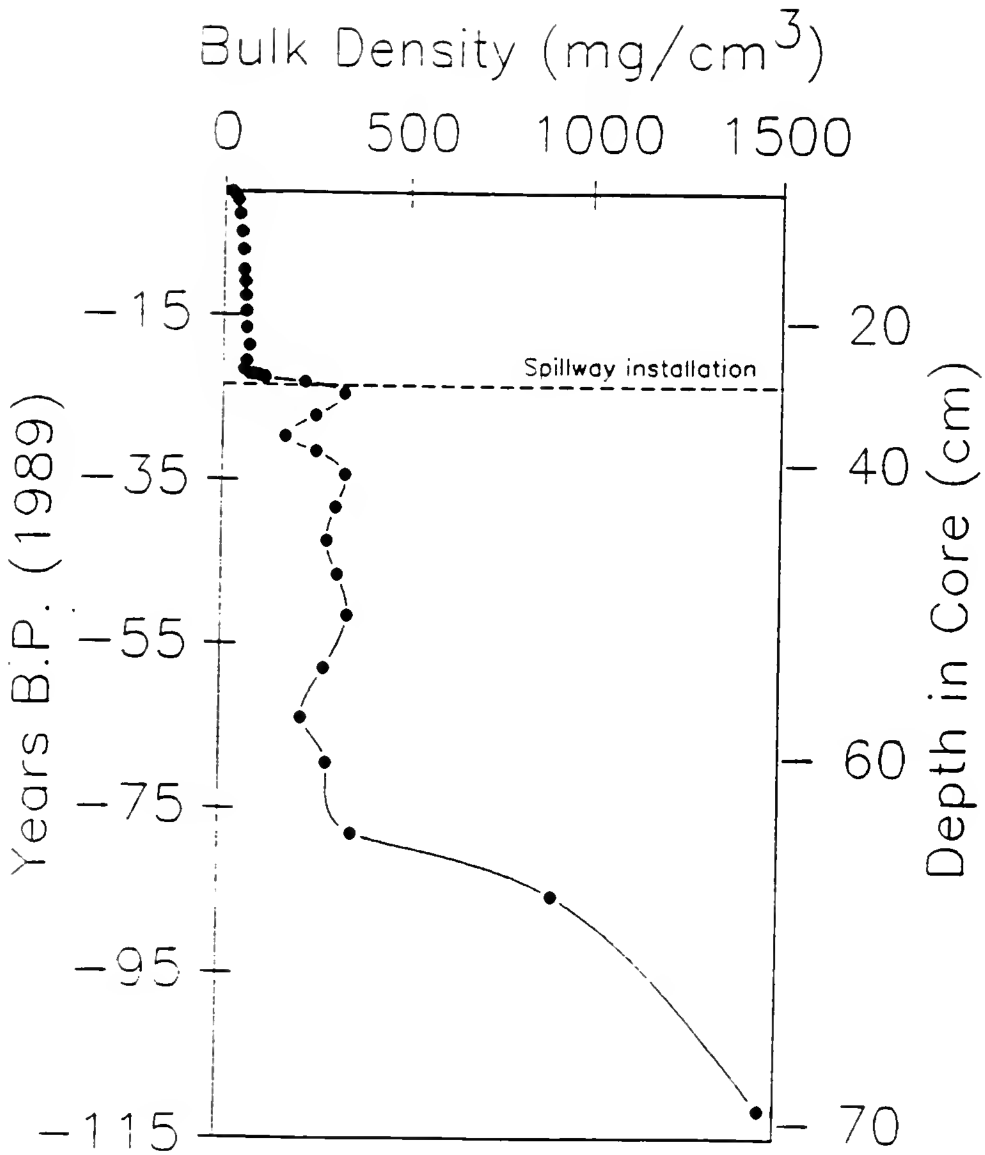


Figure 2-4. Bulk density profile for core site 8, Newnan's Lake.

subtropical lakes where high temperatures stimulate bacterial respiration in organic substrate (McDonnell and Hall 1969). In addition, resuspended sediments may contribute significant quantities of nutrients to the overlying water column through desorptive processes or entrained pore fluid (Bengtsson 1975; Cooke et al. 1977). Pollman (1983) reported a more than twofold increase in mid-lake orthophosphorus concentrations under a moderate wind event (approximately 8.9 m/s) in a central Florida lake with a mean depth and sediment composition similar to Newnan's Lake. Pronounced increases in the nitrogen and phosphorus content of surficial sediments deposited since spillway installation may enhance internal nutrient cycling and, therefore, change plant communities and lake productivity. For example, using data from Pollman's (1983) sediment-dispersion nutrient-release model, resuspension of the upper 1 cm of flocculent sediment at station 8 would cause a 70% increase in the phosphorus concentration of the overlying water through entrainment of interstitial water and desorption from suspended sediments.

Carbon:nitrogen:phosphorus ratios (C:N:P) demonstrated both temporal and spatial changes in the composition of the deposits (Table 2-1). Measurement of total carbon (TC) and total organic carbon (TOC) demonstrated that almost all sedimentary C is organic (TOC = 0.996 × TC; standard error = 0.003). Postspillway deposits were low in P compared with material deposited prior to 1967. This reduction in P content was most pronounced relative to N content at stations 6 and 1 (Table 2-1). While many factors could be considered in interpreting this difference, including changes in fractionation of sedimentary P and variations in Fe content of the sediments (Engstrom and Wright 1984), such information was not collected for this work. The reduced P content of recent sediments may be a reflection of a change toward P-limitation of primary production in the lake.

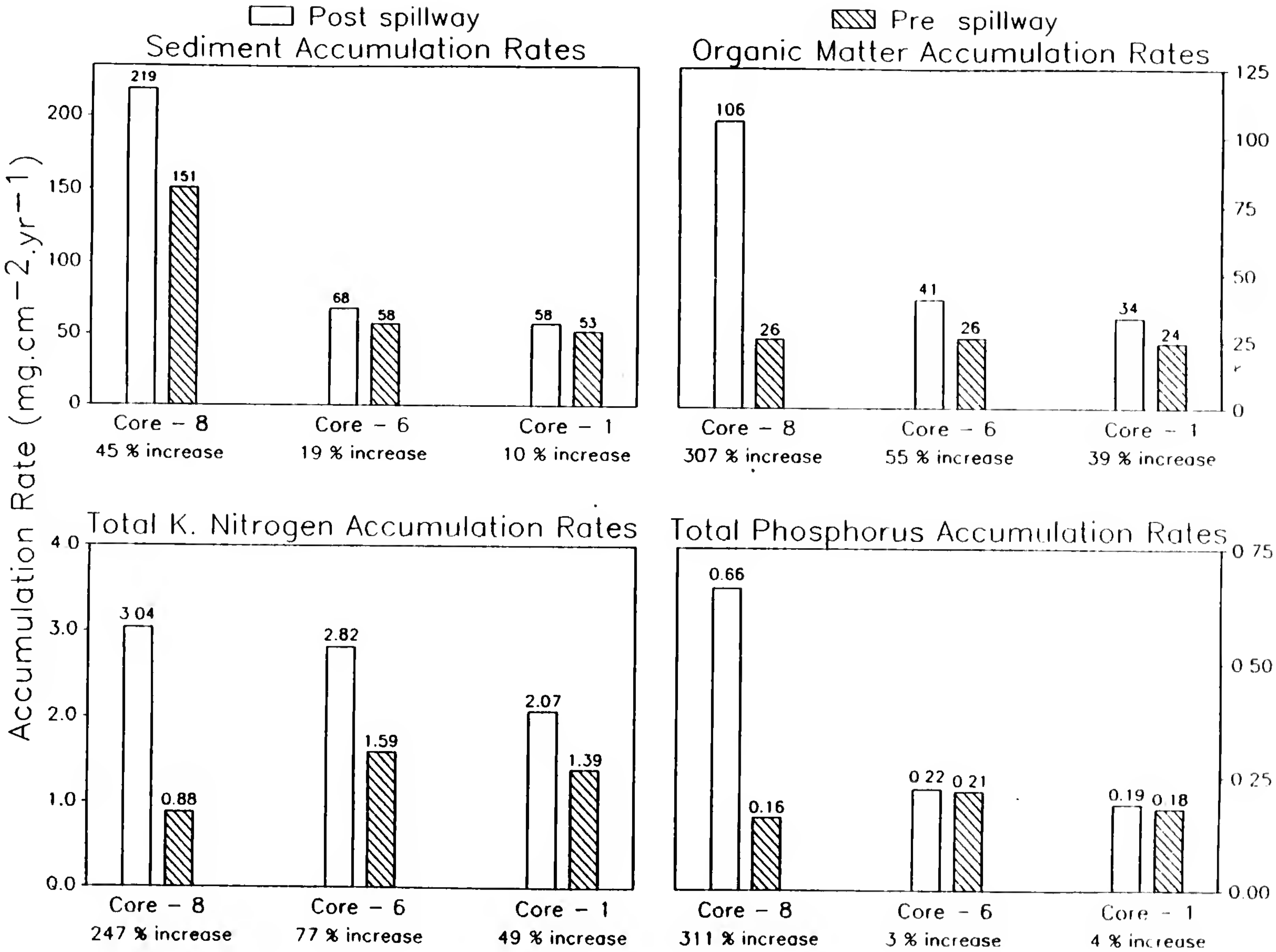
Table 2-1. Total carbon (C) to total Kjeldahl nitrogen (N) to total phosphorus (P) ratios (weight/weight) for cores 8, 6, and 1 pre- and postspillway installation, Newnan's Lake (FL).

	Core		
	8	6	1
Post C:P	82	91	91
Pre C:P	40	44	43
Post C:N	13	10	8
Pre C:N	7	6	6
Post N:P	6	10	11
Pre N:P	6	7	7

Spatially within the lake, the amount of P in the sediments relative to C and N was higher in "downstream" areas (e.g. site 8). This may result from rapid entrainment of P and reduced N:P ratios during windy conditions in shallow systems (Hamilton and Mitchell 1988) followed by downstream transport and density-dependent settling of the fine particulates with which P is associated (Engstrom and Wright 1984). High C:N ratios in core 8 deposits may reflect a somewhat higher degree of decomposition of bottom material in the lower portion of the lake. The low C:N ratios lakewide (between 6 and 13) imply an autochthonous origin of the organic matter in the sediment matrix (cf. Wetzel 1983). The contribution of non-Kjeldahl nitrogen to this ratio is small in anaerobic sediments (Reddy and Patrick 1984).

Net accumulation rates of sediment, organic matter, and nutrients have increased greatly close to the spillway since its construction (Figure 2-5). This increase was progressively less at sites farther from the spillway, where accumulation rates corresponded well with mid-lake values reported for lakes of similar trophic state (Deevey et al. 1986: Binford and Brenner 1986). The highest rates were found consistently at the station most distant from significant material inflow in the northern portion of the lake. A 45% higher sedimentation rate at station 8 amounts to an additional 11 cm of material accumulation and a 9% reduction in water depth in this shallow system during the 24 years since dam construction.

Figure 2-5. Net accumulation rates of bulk sediment, organic matter, total Kjeldahl nitrogen, and total phosphorus for core sites 8, 6, and 1, Newnan's Lake. Percent increase since spillway installation noted.


Post spillway
Pre spillway
Sediment Accumulation Rates
Organic Matter Accumulation Rates
Total K. Nitrogen Accumulation Rates
Total Phosphorus Accumulation Rates
Accumulation Rate (mg.cm−2.yr−1)
219
151
68
58
58
53
106
26
41
26
34
24
3.04
0.88
2.82
1.59
2.07
1.39
0.66
0.16
0.22
0.21
0.19
0.18
Core − 8
Core − 6
Core − 1
45 % increase
19 % increase
10 % increase
307 % increase
55 % increase
39 % increase
247 % increase
77 % increase
49 % increase
311 % increase
3 % increase
4 % increase

A pronounced increase in the accumulation rate of P at station 8 was observed (3-3.5 times the measured rate at stations 6 and 1), while recent N deposition is similar among the three stations (Figure 2-5). This "focusing" of P in sediments in the lower portion of the lake parallels the decreased C:P and N:P ratios at this downstream station. The higher rate of entrainment of P (compared to N) by wind-induced wave action (Maceina and Soballe 1990) likely contributes to P focusing in this shallow, exposed lake. Its short water residence time and obstructed outflow promote horizontal transport and subsequent accumulation close to the dam.

Other factors may have contributed to increased sedimentation, but their effect would have resulted in changed sedimentation rates lake-wide, rather than concentrated toward the spillway. These factors include herbicide treatment of aquatic macrophytes and added nutrient runoff from the watershed. Between 1972 and 1977 a total of 130 ha (i.e. 4%) of lake surface area was treated chemically to reduce the abundance of *Eichhornia crassipes* (water hyacinth) and *Hydrilla verticillata* (hydrilla). In November of 1988 an additional 110 ha received chemical treatment (Hinkle, Florida Department of Natural Resources, pers. comm.). Analysis of aerial photographs (U.S. Geological Survey 1966, 1988) did not show significant land-use changes in the basin during the last 22 years. The number of residences less than 1 km from the Newnan's Lake shoreline increased from 16.0 to 19.2 units/km^2 between 1966 and 1988.

Summary and Conclusions

The Newnan's Lake spillway, intended to stabilize lake stage, changed material transfer between water and sediment. This change, operating over a period of decades, can be quantified using paleolimnological techniques. A transect of ^{210}Pb dated profiles perpendicular to the dam at the lake outlet demonstrated a distinct impact from its installation in 1967 on net rate of material accumulation and sediment composition. While net accumulation rate of bulk sediment was only 45% higher, resulting in a 9% reduction in water depth, rates for organic matter and nutrients tripled compared to prespillway conditions. P accumulation was particularly enhanced close to the spillway compared with other stations. If allowed to persist, increased material transfer between water and sediments may produce loss of lake volume and changes in benthic habitat. The threefold increase in the trapping of nutrients may change plant communities and lake productivity.

CHAPTER 3
REMOVAL OF PARTICULATE ORGANIC MATTER/NUTRIENTS AND OXIDATION/CONSOLIDATION OF EXPOSED LITTORAL SUBSTRATE DURING A SHORT-TERM DRAWDOWN

A three-month gravity drawdown of the lake was initiated on April 24, 1989 by removing all spillway flashboards. The objectives of this drawdown were to improve fish growth and recruitment and to flush nutrient-rich detritus from the lake. Accumulation of flocculent and nutrient-rich sediments in the lake (Chapter 2) and the presence of this flocculent substrate in the littoral zone may reduce the availability of preferred nesting habitat for centrarchids, such as the largemouth bass (*Micropterus salmoides*) and black crappie (*Pomoxis nigromaculatus*) (Eddy and Underhill 1978; Bruno 1984). Exposing the littoral zone lake-bottom during drawdown may allow oxidation and consolidation of this substrate, which may improve this habitat for future fish spawning. Seed germination and firm substrate promote the establishment of littoral plant communities, which may provide refuge and feeding habitat and promote fish recruitment.

Research on other Florida lakes has quantified effects of drawdowns on fish populations (Wegener and Williams

1974), aquatic invertebrates (Wegener et al. 1974), and littoral plant communities (Holcomb and Wegener 1971; Goodrick and Milleson 1974; Hestand and Carter 1975). However, no information is available on the impact of drawdown practices on the flushing of organic matter and nutrients from Florida lakes. The objectives of this part of the study were: (1) To measure the flushing of organic matter, nitrogen and phosphorus in surface discharge from Newnan's Lake prior to and during the gravity drawdown and (2) to determine the effect of exposing littoral zone sediments to air on oxidative removal of organic matter and long-term consolidation of the substrate.

Materials and Methods

Discharge, Rainfall, and Lake Stage

Discharge through Prairie Creek, rainfall, and lake water-level were measured starting February 20, 1989, 9 weeks prior to drawdown, and continued throughout the drawdown period until July 31, 1989, 2 weeks after the spillway boards were re-installed.

Surface flow from the lake was measured weekly using an Ott flow-meter (Type C2-10150) approximately 30 m downstream from the dam in Prairie Creek (Figure 3-1). The frequency of sampling was increased during storm events to include a minimum of 1 prestorm flow, 1 peak flow, and 1 poststorm flow measurement. The creek width was divided in 1 m subsections, and flow was measured in each section at

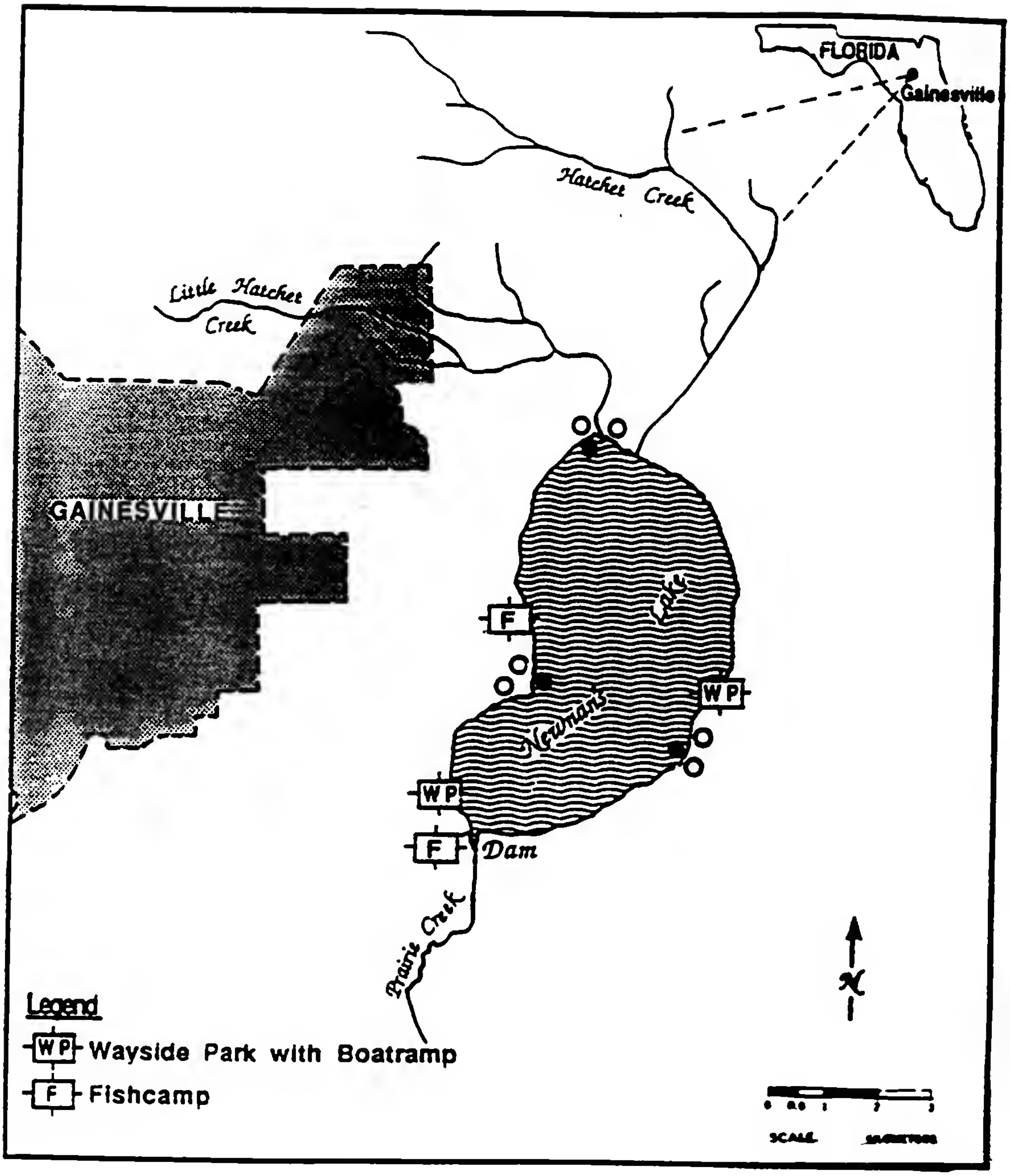


Figure 3-1. Map of Florida showing study area location and map of Newnan's Lake. Open circles indicate the locations of field enclosures, and closed circles indicate the sediment collection sites for the oxidation/consolidation laboratory experiments.

0.6 depth (when channel depth < 30 cm) or at 0.2 and 0.8 depth and averaged (when channel depth > 30 cm). Discharge was computed by multiplying flow in each subsection by its cross-sectional area. Total discharge equals the sum of all subsection discharges. If the measured discharge differed from the discharge computed using USGS stage-discharge ratings by more than 10%, then discharge was remeasured and averaged with the first measurement. A calibration flume (Department of Civil Engineering, University of Florida) was used to determine the accuracy of the Ott flow-meter.

The Gainesville Flight Service Station (National Oceanic and Atmospheric Administration) kept daily rainfall records during the study period using a standard (20.32 cm diameter) U.S. Weather Service raingauge, located near the center of the watershed. Lake levels were recorded daily by the U.S. Geological Survey (station # 02240900) located on the west side of the lake.

Water Quality Analysis

Replicate water samples were collected concurrently with discharge measurements 20 m upstream from the spillway in the center of the creek. Water samples were collected in acid washed Nalgene containers, immediately stored on ice, and frozen within 1 hour of collection. Samples were analyzed for total suspended solids (TSS), particulate organic matter (POM), total Kjeldahl nitrogen (TKN), and total phosphorus (TP). The analyses were completed within 90 days of collection.

Analysis for TSS was according to Standard Methods (method 209C, A.P.H.A. 1985) using pre-weighed and pre-muffled glass fiber filters (Whatman, 934-AH, 4.25 cm diameter) with an effective particle retention of 1.5 μm. Replicate analyses were done on 0.5 liter samples. POM was measured as weight loss on ignition (at 550°C for 1 hour, followed by rehydration with distilled water and drying at 95°C for 24 hours) (method 209D, A.P.H.A. 1985) of oven-dry samples (95°C for 24 hours). TKN analyses were according to E.P.A. method 351.2 (E.P.A. 1979) using a 40-cell block digestor and a Technicon II semi-automated manifold. The digested sample was also used for TP determination. The liberated orthophosphate in the digested samples was determined using the ascorbic acid method (method 424F, A.P.H.A. 1985). Absorbance of the samples was read at 880 nm on a Bausch and Lomb spectrophotometer (Model 21) with a light path of 2.5 cm. Both nitrogen and phosphorus analyses were done on replicate samples and averaged.

Sediment Oxidation

Oxidation rates of organic matter in exposed littoral zone sediment were measured in the laboratory and the field. Littoral surface sediment was collected from 3 stations in the lake (Figure 3-1) and homogenized in a blender in the laboratory. Subsamples (n=25) of 100 ml each were incubated in crucibles at room temperature for 78 days under 3 treatments. One set was allowed to dry completely and remained dry during the test period; one set was kept wet without standing water by adding up to 8 ml of distilled water per week, and a third set was kept inundated (with 2-3 cm of distilled water). The duration of this experiment (78 days) was chosen to coincide with the expected length of time that littoral substrate would be exposed in the field. Crucibles in the laboratory were rotated every 48 hours to promote equal exposure to light conditions (generally between 20-30 Lux, measured with a Li-Cor photometer, Model LI 188). Organic matter content before and after incubation was determined using weight loss on ignition (at 550°C for 1 hour followed by rehydration with distilled water and drying at 95°C for 24 hours) of oven-dry samples (95°C for 24 hours).

PVC enclosures (surface area = 180 cm^2) were installed in pairs at 3 locations in exposed littoral zone sediments (Figure 3-1). The enclosures were pushed through the top layer of wet organic substrate to a point 5-10 cm into the underlying sand. Enclosed sediment was shielded from precipitation by an elevated plexiglass roof. This design prevented removal of enclosed substrate by water and/or wind erosion. Hardware cloth (1.25 cm mesh size) was put on top of the enclosure to deter animal disturbance.

Organic matter content at the beginning of exposure was quantified by average weight loss on ignition of 3 "core" samples adjacent to the enclosure. Each sample covered an area equivalent to the enclosure and was analyzed in its entirety down to sand. After 28 days of incubation, the organic matter content of the enclosed substrate down to sand was determined. Net oxidative removal (g $m^{-2}day^{-1}$) was computed as the difference between pre- and postincubation measurements.

Sediment Compaction

A small-scale sediment compaction study was also performed to evaluate changes in substrate bulk density, water content and percent organic matter at various times after exposure and after reflooding. Four vessels (glass, 1 liter) containing 500 ml of homogenized littoral substrate were allowed to dry in the laboratory for 51 days, then inundated (to a depth of 10 cm) for 149 days, simulating the water regime in the field. Incubation of the substrate occurred at room temperature, indoor light conditions, and approximately 40% relative humidity. The locations from which sediment samples were collected are shown in Figure 3-1. Only the top 5 cm of substrate was used. Ten water content (percent) and organic matter content (percent) determinations were made at 5 times during exposure, at the time of reflooding, and 4 times during reflooding.

Results and Discussion

Discharge, Rainfall, and Lake Stage

During the 3.5 months prior to drawdown lake stage dropped from 20.33 m (MSL) to 20.07 m (MSL) due to lack of rain. The boards in the spillway were removed on April 24 (1989) after which the water level in the lake dropped from 20.07 m (MSL) to 19.80 m (MSL) in 1 month (Figure 3-2). Lake stage remained between 19.70 and 19.80 m MSL for 8 weeks. This stage was the lowest recorded during the last 25 years, with the exception of a brief period during the drought of 1981 (when the lake reached 19.65 m MSL). The maximum drop in water level during drawdown was 36 cm. Severe lack of rain during the months prior to drawdown (Table 3-1) resulted in lake water-levels approximately 41 cm below average for early spring. This reduced the amplitude of the drop in lake water-level. Small elevation gradients in the basin and the gradual build-up of obstructions in the lake outflow upstream and downstream from the spillway since its construction (Crider 1972; personal observation) likely prevented a more dramatic drawdown of water level. Those obstructions to the flow in Prairie Creek may have resulted from the impact of the spillway on stage level and discharge volumes in the creek. Low-flow and low-stage conditions downstream from the spillway are much reduced compared to the natural hydroperiod of the creek prior to dam construction in 1967 (Gottgens 1987).

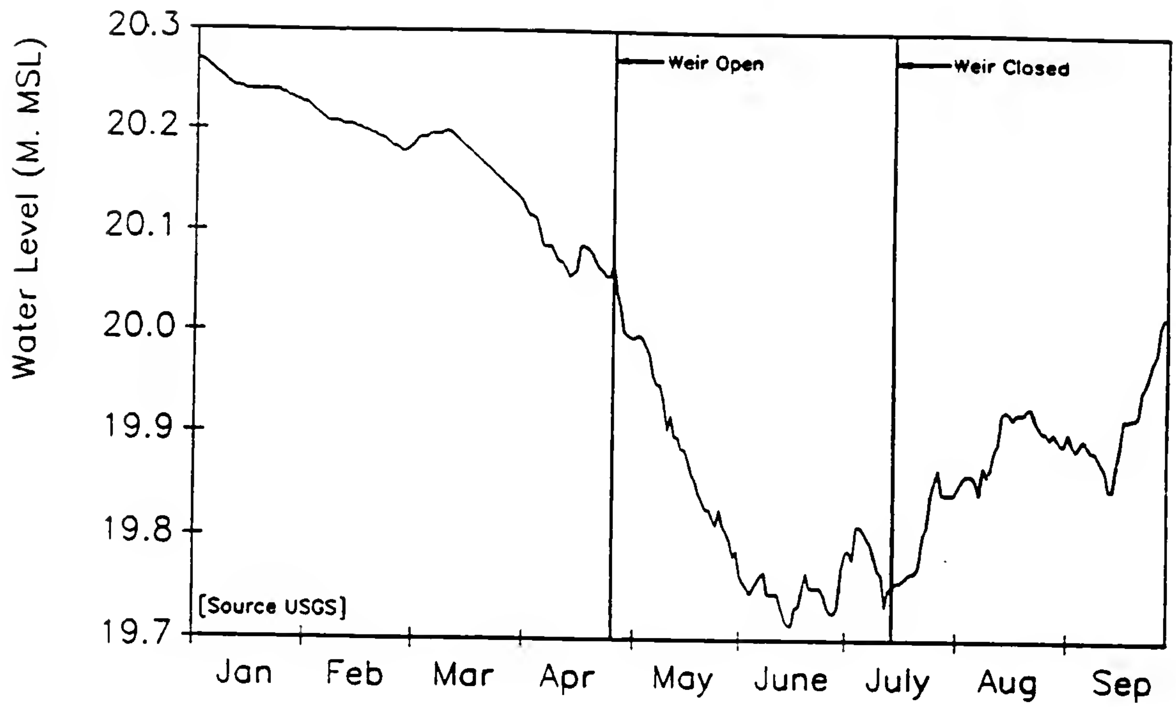


Figure 3-2. Water level record for Newnan's Lake for the period of January-September 1989 [Source: U.S. Geological Survey station # 02240900].

Table 3-1. Average rainfall (1897-1987) and rainfall during first 7 months of 1989. Records given in cm.

	Jan	Feb	Mar	Apr	May	Jun	Jul	Total
Average	7.7	9.0	9.1	7.6	8.9	16.7	18.2	77.2
1989	2.6	3.0	5.5	7.4	5.2	24.4	10.0	58.1
Deficit	5.1	6.0	3.6	0.2	3.7	-7.7	8.2	19.1

[Source: National Oceanic and Atmospheric Administration, Gainesville Flight Service Station, Florida]

Assuming a reduction of the mean water depth from 1.5 m to 1.2 m during the drawdown, it is estimated that 20% of water volume of the lake was removed during the drawdown period. Assuming unaltered base-discharge rates from the lake through Prairie Creek during the drawdown period, integration of the discharge curve (Figure 3-3) and then subtracting base-discharge yields an estimate of 1.75×10^6 m^3 of water flushed from the lake by removal of the spillway. Base-discharge is defined as the discharge flowing through the channel with the spillway in place, and is estimated by measuring average discharge prior to dam removal and after re-installation of the flashboards. In both cases an averaging period of 2 weeks was used. Below average rainfall during the drawdown period (Table 3-1) and high evapo-transpiration contributed considerably to the decrease in lake stage. Discharge through Prairie Creek dropped to approximately twice the assumed base-discharge within 6 weeks after spillway removal and remained at that

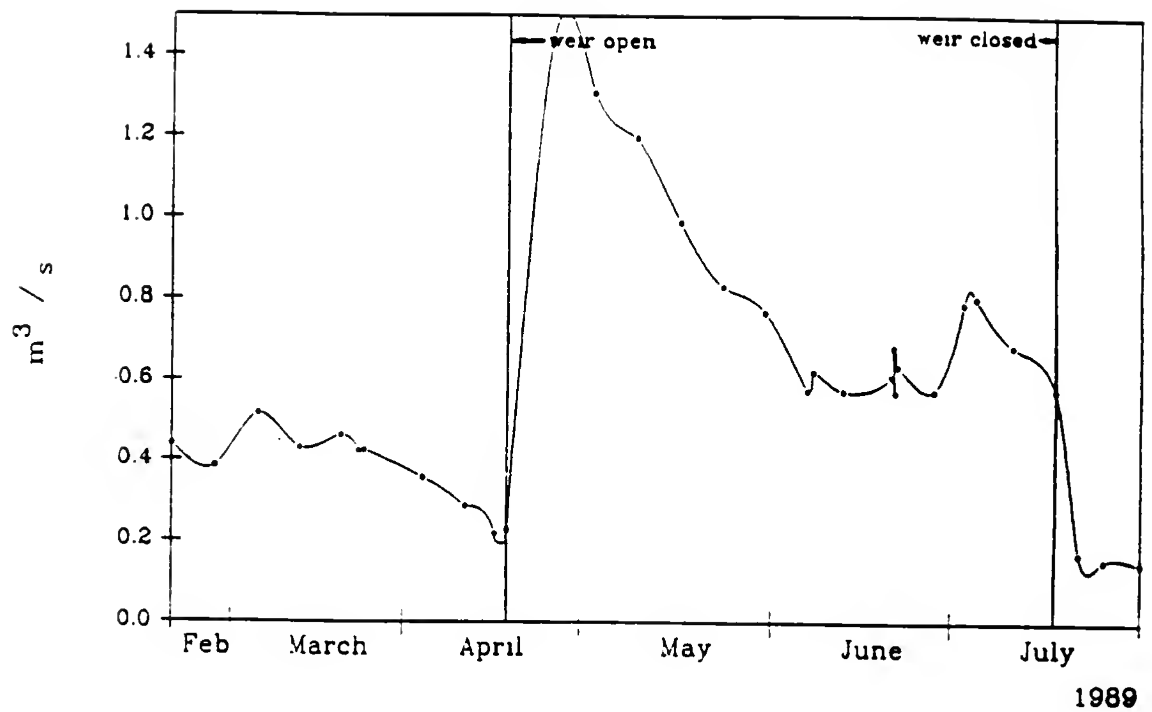


Figure 3-3. Discharge (m^3/sec) through Prairie Creek @ Florida S.R. 20 for the period of February-July 1989.

rate until closing of the weir (Figure 3-3). In spite of low total rainfall, 5 storm events of low to moderate intensity were included in the sampling period(Table 3-2).

Table 3-2. Precipitation and wind conditions during sampled storm events, Newnan's Lake.

	Dates (1989)	Amount* (cm)	Wind conditions* (km/hr)
Pre-drawdown	March 22-23	1.09	0-18 from N
	April 14-15	2.03	0-12 from N or W
During drawdown	May 29	2.26	0-40 misc. dir.
	June 18-19	4.52	0-16 misc. dir.
	July 16-17	1.42	0-16 misc. dir.

* Measured at spillway with standard 2.54 cm glass raingage and Dwyer handheld windmeter

Removal of Particulate Matter and Nutrients

Increased flow through Prairie Creek after spillway removal increased discharges (kg/day) of particulate organic matter (POM), total Kjeldahl nitrogen (TKN), and total phosphorus (TP) (Figure 3-4a,b, and c). POM was a consistent fraction of total suspended solids during the sampling period (POM = 0.76 × TSS; N=64; R^2=0.96). Assuming base-discharge rates (kg/day) through Prairie Creek during the drawdown period, integration of total discharge rates and subtraction of base-discharge rates yields an estimate of the amount (kg) of POM, TKN, and TP flushed from the lake owing to drawdown (Table 3-3).

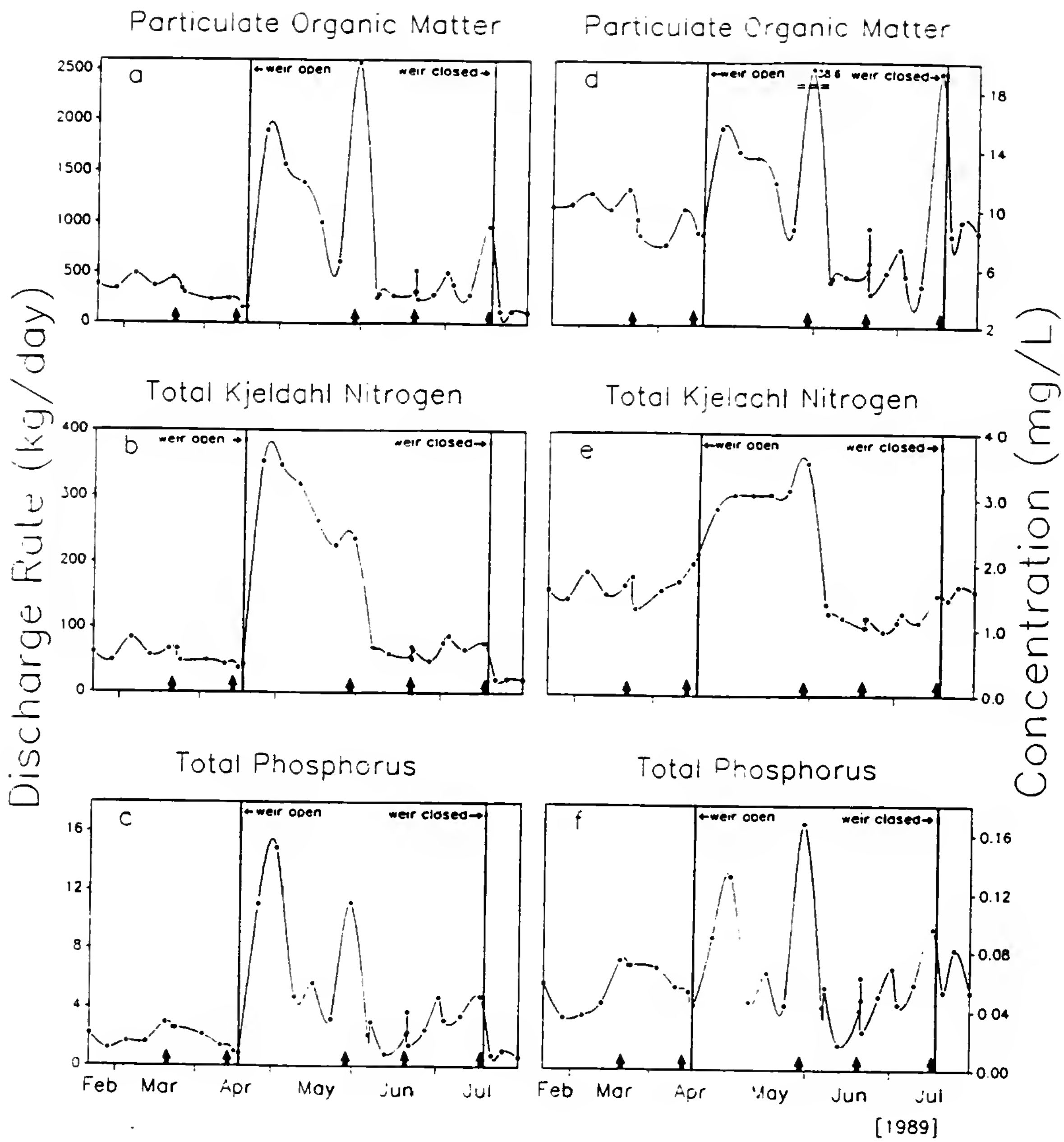


Figure 3-4. Characteristics of the discharge from Newnan's Lake through Prairie Creek. Opening and closing of spillway indicated. Discharge rate (kg/day) of POM, TKN, and TP (Figure 3-4a, b, and c). Concentration (mg/l) of POM, TKN, and TP in discharge (Figure 3-4d, e, and f). Arrows indicate sampled storm events.

Table 3-3. Amounts of total suspended solids, particulate organic matter, total Kjeldahl nitrogen, and total phosphorus flushed in excess of base-discharge from Newnan's Lake during partial drawdown.

	kg dry weight removed by drawdown	mg/m^2 lake area removed by drawdown
Total suspended solids	59,247	2,043
Particulate organic matter	46,537	1,605
Total Kjeldahl nitrogen	8,840	305
Total phosphorus	290	10

Computed on a m^2-basis, removal during drawdown is small compared to the likely stores of flocculent sediment in the lake, with an average estimated thickness of 70 cm (Skoglund 1990). Two factors may account for the relatively low removal rates of particulate material from the basin. First, the lower incidence of storm events during the period of drawdown may have precluded considerable resuspension of flocculent bottom material (Pollman 1983). Wind-induced wave action will stir the water column and may promote flushing of resuspended matter (Maceina and Soballe 1990). Second, the low water-level at the onset of drawdown results in small hydraulic head and discharge rates. Low discharge rates depress removal of resuspended material.

Concentrations of POM, TKN, and TP in the lake discharge are shown in Figure 3-4d, e, and f. The data points represent the average value of two replicate samples. "Error bars" averaged ±0.40 mg/l for POM (range 0-1.63), ±0.15 mg/l for TKN (range 0-0.42), and ±0.01 mg/l for TP (range 0-0.03). The concentrations in the discharge were higher during the first month of drawdown compared with the pre-drawdown period (Figure 3-4d, e, and f). These differences are statistically significant ($P<0.05$, $P<0.05$, $P<0.01$, respectively) using a pooled analysis of variance (Byrkit 1975) and assuming no autocorrelation among data points (Table 3-4). After this period, when discharge decreases to about twice the assumed base-discharge (see Figure 3-3) concentrations of POM, TKN, and TP drop to near predrawdown levels. At these low lake stages, the outflow from Newnan's Lake may be restricted to the less-turbid surface layer of water released through the shallow creek.

Table 3-4. Concentrations of particulate organic matter (POM in mg/l), total Kjeldahl nitrogen (TKN in mg/l), and total phosphorus (TP in µg/l) in surface discharge from Newnan's Lake prior to and during the first month of drawdown. Number given are the mean, number of measurements (N), and standard error of the mean (S.E.)

	Predrawdown			During 1st month of drawdown		
	Mean	N	S.E.	Mean	N	S.E.
POM	9.48	22	0.25	13.61	8	0.47
TKN	1.72	22	0.05	3.02	8	0.03
TP	55.81	22	2.88	83.13	8	11.42

Considerable changes in water quality of the outflow occurred during the sampled storm events. Three sampled storm events during the drawdown period produced increases in the concentration of POM, TKN, and TP (Figure 3-4d, e, and f). Other peaks in these time patterns may have been associated with wind events which were not sampled. Although a more detailed analysis (incorporating daily wind data and in-lake water quality) would provide better evidence, it appears that sampled storms were effective in resuspending surficial flocculent sediment in Newnan's Lake. This corresponds with findings in other shallow lake systems (Sheng and Lick 1979; Somlyódy 1982). Mixing of these deposits in the water column enhances their removal through flushing. The lack of high-intensity wind events (e.g. in excess of 30 km/hr) during the study period contributed to the low particle flushing rates encountered. Two storms of low intensity, sampled prior to removal of the spillway, produced small or no increases in particulate matter and nutrient concentrations of the flow through the section of the dam with the missing top-flashboard. Evidently, storms are effective in flushing particulates and nutrients from the system. The flushing, however, is typically significant when obstructions such as the spillway are removed from the lake outlet.

Oxidation of Organic Matter in Exposed Littoral Sediments

A second objective of the short-term drawdown of Newnan's Lake was to improve littoral zone habitat. Exposure of littoral zone lake bottom to air may allow oxidative removal of organic material (Wegener et al. 1974) and should consolidate flocculent sediments (Fox et al. 1977). This improves the area for future sportfish spawning and provides firm rooting for aquatic macrophytes (Holcomb and Wegener 1971), which may contribute to a higher standing crop of aquatic macro-invertebrates and, eventually, fish (Wegener and Williams 1974). A vegetated littoral zone may function as an effective nutrient trap and reduce nutrient input from runoff into the lake (Mickle and Wetzel 1978). Numerous authors have suggested that aquatic macrophytes can inhibit the development of algae (Canfield et al. 1984; Crisman 1986b). Accumulation of algae are perceived as a persistent problem in Newnan's Lake.

No significant net oxidation of littoral substrate was noted after 28 days of *in situ* exposure (Figure 3-5). The difference in organic matter content between pre- and postexposure enclosures was statistically not significant (paired t-test, $P=0.05$). Field observations showed production of organic matter in the form of germinating seeds, roots, and above-ground plant biomass inside the enclosures. If oxidation of the sediments occurred, it may

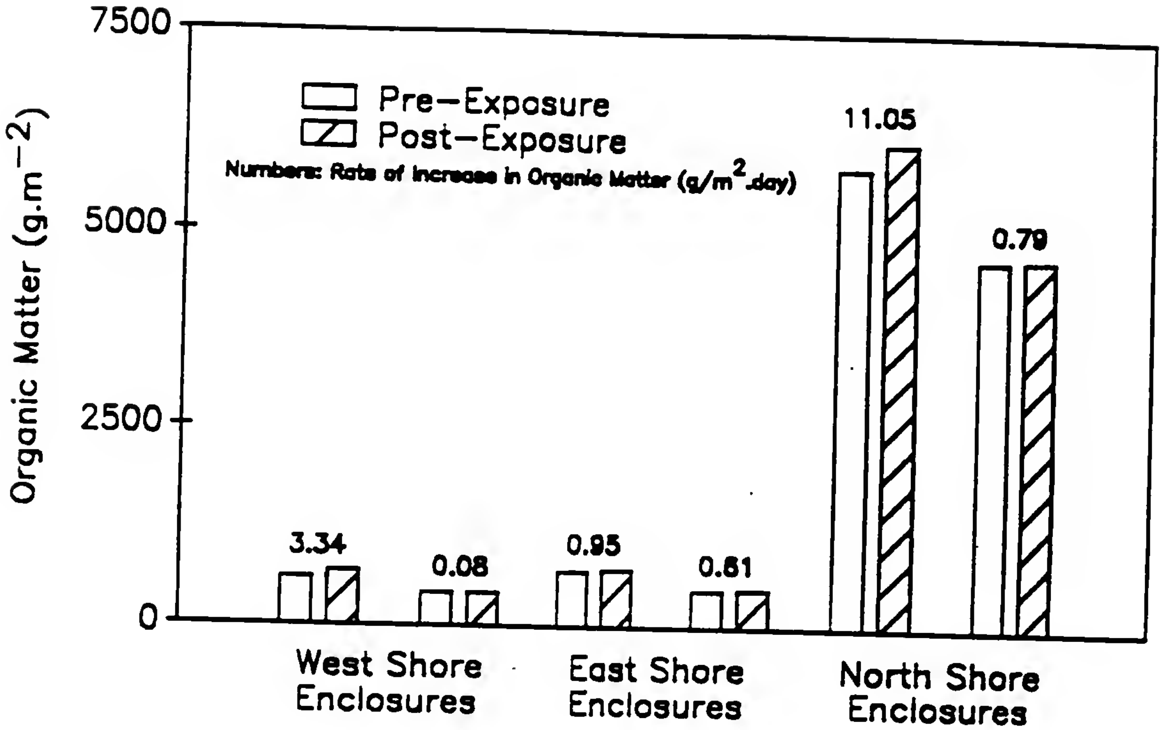


Figure 3-5. Organic matter content (g/m^2) at the onset ("pre") and after 28 days of exposure ("post"), and rate of increase in organic matter ($g\ m^{-2}d^{-1}$) in Newnan's Lake littoral sediments (field data).

have been masked by this organic matter production. It is possible that oxidation rates may have been quite low, particularly if the organic sediments were largely humic compounds, which are relatively resistant to microbial degradation (Sederholm et al. 1973).

The laboratory experiments did not provide evidence of oxidation of these sediments either. Oxidation rates were extremely low and not significantly different between dry, moist, and permanently inundated substrate (Table 3-5).

Table 3-5. Mean oxidation rates of incubated littoral sediments in the laboratory under 3 different treatments.

	N	Mean oxidation rate ($mg\ g^{-1}d^{-1}$)	S.D.	Range
Inundated	9	0.23	0.05	0.17-0.30
Moist	9	0.24	0.05	0.16-0.35
Dry	7	0.19	0.05	0.13-0.26

Production of organic matter in these laboratory chambers under the low-light regime was less likely than in the field enclosures. Hence, these experiments suggest that oxidation rates were indeed low. These results support findings by Fox et al. (1977), who noted no significant decomposition of organic matter in sediment from Lake Apopka, Florida, using a series of laboratory experiments.

Consolidation of Littoral Sediments

In the laboratory consolidation-experiment, water content of exposed sediments decreased from 90 to 50 percent (Figure 3-6) and remained moderately compact 149 days after reflooding. Organic matter content remained constant throughout the duration of the experiment (Figure 3-6). Firmer substrate will reduce rates of resuspension during periods of high winds.

Summary and Conclusions

The drought of 1989 produced a low lake-stage (41 cm below average) at the start of the drawdown period and a low incidence of storm events during the drawdown. This reduced the discharge rates of particulate organic matter and nutrients from the lake.

Concentrations of POM, TKN, and TP in the lake discharge were significantly higher during the first month of drawdown than during the pre-drawdown period. These concentrations dropped to near predrawdown levels at lower lake stages, when the sill depth at the mouth of Prairie Creek may have restricted the outflow from the lake to the less-turbid surface layer of water.

Storm events produced increases in concentration of particulate organic matter and nutrients in the discharge from the lake during drawdown. Storms sampled prior to opening of the spillway did not cause such increases. Storms resuspended fine particulate deposits in this

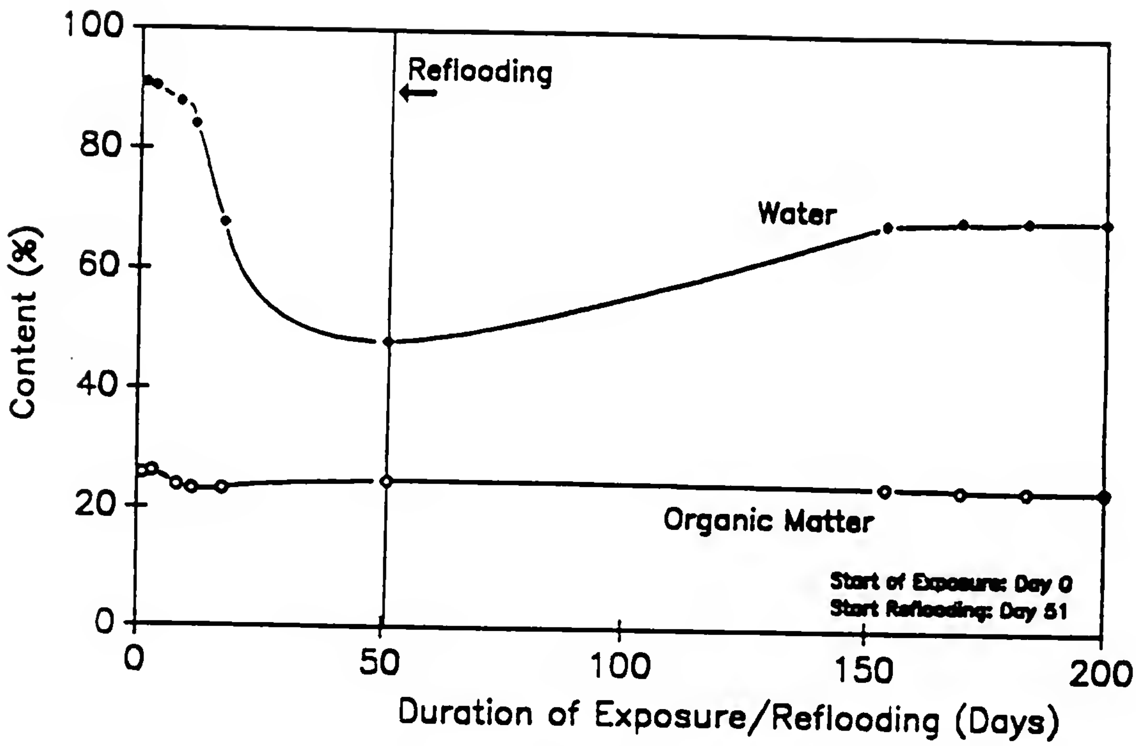


Figure 3-6. Changes in water and organic matter content (weight/weight ratio) in Newnan's Lake littoral sediments upon exposure and after reflooding in a laboratory setting.

shallow, exposed lake and promoted flushing of this material.

The sill at the mouth of Prairie Creek reduces the likelihood of a gravity drawdown to a stage much lower than that accomplished (19.70 m MSL). At this stage, the water level is barely lakeward of the cypress tree fringe and drying/consolidation of lake bottom is limited to a narrow littoral zone fringe.

Field and laboratory tests did not show oxidative removal of organic matter from exposed areas of the lake bottom. Consolidated sediments remained moderately firm after reflooding in a laboratory experiment and may, therefore, provide improved substrate for rooted aquatic vegetation.

CHAPTER 4
REDISTRIBUTION OF ORGANIC SEDIMENTS FOLLOWING A SHORT-TERM DRAWDOWN.

Water-level drawdown is a well established lake management technique. It has been used to influence the abundance and composition of aquatic plant communities (Holcomb and Wegener 1971; Hestand and Carter 1975; Cooke 1980; Tarver 1980), increase fish standing crop (Lantz et al. 1964; Wegener and Williams 1974), and consolidate littoral sediment (Holcomb et al. 1975; McKinney and Coleman 1980). Drawdowns have also produced higher densities of littoral macroinvertebrates (Wegener et al. 1974), increased nutrient concentrations in the water column (Serruya and Pollingher 1977), and reduced dissolved oxygen levels by disrupting the thermal stratification of the water column (Richardson 1975). The intent of the 1989 short-term drawdown in Newnan's Lake was to improve littoral habitat for fish growth and recruitment by allowing oxidation and consolidation of exposed littoral zone lake-bottom (Chapter 3).

In spite of the frequent use of water-level drawdown, little is known about its impact on erosion of littoral sediments. Low water levels during drawdown increase wind-wave action on littoral substrate. The resulting physical resuspension of bottom material may then be followed by transport and gravitational settling of the entrained particles in deeper areas (Bengtsson et al. 1990). As such, material can be focused from the littoral zone to profundal substrate (Davis 1968). This process may be particularly pronounced in lakes with fine, organic bottom substrate, i.e. those where water-level drawdown is most commonly applied.

The effects of enhanced resuspension of bottom material may include increased turbidity and reduced light penetration in the water column. It may also exert considerable oxygen demand (James 1974) and lead to increased availability of nutrients in the overlying water for algal utilization (Holdren and Armstrong 1980; Pollman 1983). "Sloughing" of eroded material to deeper parts of the lake reduces maximum water depth and may lead to the development of substantial shallow areas. This increases the potential of the lake ecosystem to support extensive growth of rooted macrophytes.

The objective of this part of the study was to quantify the removal of flocculent sediments from the littoral zone and determine their redistribution to deeper areas of the lake during a short-term drawdown of Newnan's Lake.

Materials and Methods

Sediment Cores

Removal and redistribution of lake bottom material due to drawdown was investigated using two series of nine sediment cores. Marker horizons were identified in the cores either from profiles of bulk density, unsupported ^{210}Pb, ^{137}Cs, organic matter and nutrient content or from direct field evidence of distinct stratigraphy. Gain or loss of sediment at each site was then quantified by matching marker horizons between pre- and postdrawdown cores.

Figure 4-1 shows the location of the cores. An electronic long-range navigation system (LORAN, Si-Tex 797) was used to ensure agreement in location of pre- and postdrawdown sampling sites. Additionally, triangular compass measurements with permanent landmarks were used for the littoral cores. Specifics of the core locations are given in Table 4-1. Latitude and longitude records were relative to the calibration site located near the southwestern boat ramp (Figure 4-1).

The accuracy of this LORAN to return to a sampling site is limited to a range of approximately 20 meters. This inherently results in error when comparing pre- and postdrawdown sediment stratigraphy. This error is reduced when in-lake variability between nearby sites, in terms of water depth and bottom stratigraphy, is low as in Newnan's

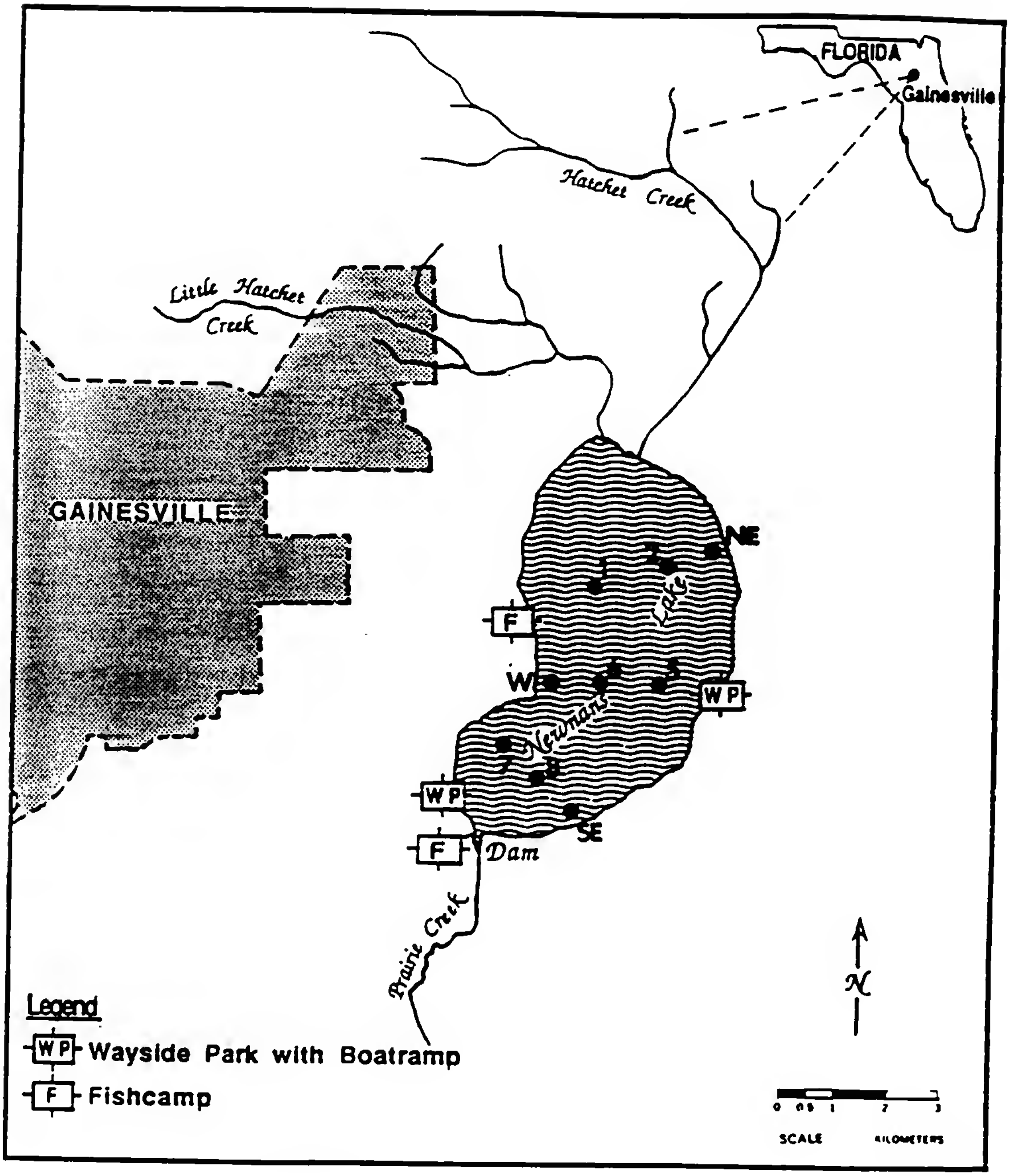


Figure 4-1. Map of Newnan's Lake with core locations indicated (filled circles).

Lake with its rather flat bottom topography (Holly 1976) and homogeneous deposits of soft sediment (Skoglund 1990). By matching several marker horizons between cores, rather than one single horizon, error may be reduced further.

Table 4-1. Newnan's Lake cores: Water depth and location.

Core	Depth (cm)	Latitude	Longitude
1-11	150	29 39 03	82 13 82
2-12	200	29 39 12	82 12 87
5-15	170	29 38 02	82 12 71
6-16	150	29 38 04	82 13 47
7-17	118	29 37 69	82 14 74
8-18	115	29 36 80	82 14 50
NE	65	29 39 27	82 12 05
SE	62	29 36 68	82 14 43
W	58	29 38 16	82 14 22

Note: Measured with Si-Tex 797 LORAN in degrees, minutes, and (minutes/100). Calibration marker at southwest boat ramp: 29 37 07/82 15 23.

Water depth at all stations was estimated by carefully lowering a Secchi disk until contact between the disk and soft bottom substrate was noticed.

Cores were collected and preserved as described in Chapter 2. Visual observations of the cores were made in the field and again during sectioning in the laboratory to detect changes in sediment color or texture which could serve as marker horizons.

Bulk Density and Nutrient Analyses

The cores were sectioned in 1 cm intervals to a depth of 30 cm, into 2 cm intervals from 30 to 80 cm depth, and into 4 cm intervals below that. The sections were sealed in plastic bags and stored in a refrigerator. Bulk density, organic matter content, total Kjeldahl nitrogen (TKN), and total phosphorus (TP) were determined as described in Chapter 2. Measurements were made at intervals selected to give a representation of the entire core profile.

Radio-isotope Analyses

Marker horizons from measurement of ^{210}Pb and ^{137}Cs levels throughout the core profiles may aid in a determination of sediment redistribution during drawdown. ^{210}Pb profiles have been reliable in documenting sediment removal due to major storm surges (Robbins et al. 1978). Pennington (1981) recorded variations in ^{137}Cs profiles in a shallow lake resulting from episodic sediment redistribution and deposition. Bengtsson et al. (1990) successfully used settling sediment traps to investigate redistribution of fine sediments in Swedish lakes. Maintaining suspended sediment traps in the water column, however, was not an option in a public use lake, such as Newnan's. Use of ^{210}Pb and ^{137}Cs in the determination of depositional markers has additional significant benefits, because such profiles permit the calculation of the age of deposited material in the cores (Eakins and Morrison 1978; Appleby and Oldfield 1983).

Between 8 and 16 ^{210}Pb and ^{137}Cs measurements were made depending on the length of the core. ^{210}Pb and ^{137}Cs concentrations were measured by direct γ-assay as described in Chapter 2.

Results and Discussion

Littoral Cores

Two series of three littoral cores were taken; northeast, southeast and west cores (Figure 4-1). Predrawdown cores were taken when water depth averaged 62 cm. Following the drawdown and natural rise of the water level to predrawdown stage the second series of three cores were taken at the same locations. Water level at these sites was lowered from 62 to 30 cm for 8 weeks during the drawdown. Sand underlying flocculent brown substrate served as a suitable marker horizon.

Sediment depth to sand decreased by an average of 42% after drawdown (Figure 4-2a), and bulk density of the remaining substrate increased by an average of 250% at all three sites (Figure 4-2b). Organic matter, TKN, and TP (expressed as weight/weight ratios) decreased at two of the three littoral zone sample sites following drawdown (Figure 4-2c, d, and e). Analogously, the amount of organic matter overlying the sandy littoral bottom was reduced (Figure 4-2f). Calculated removal rates are given in Table 4-2.

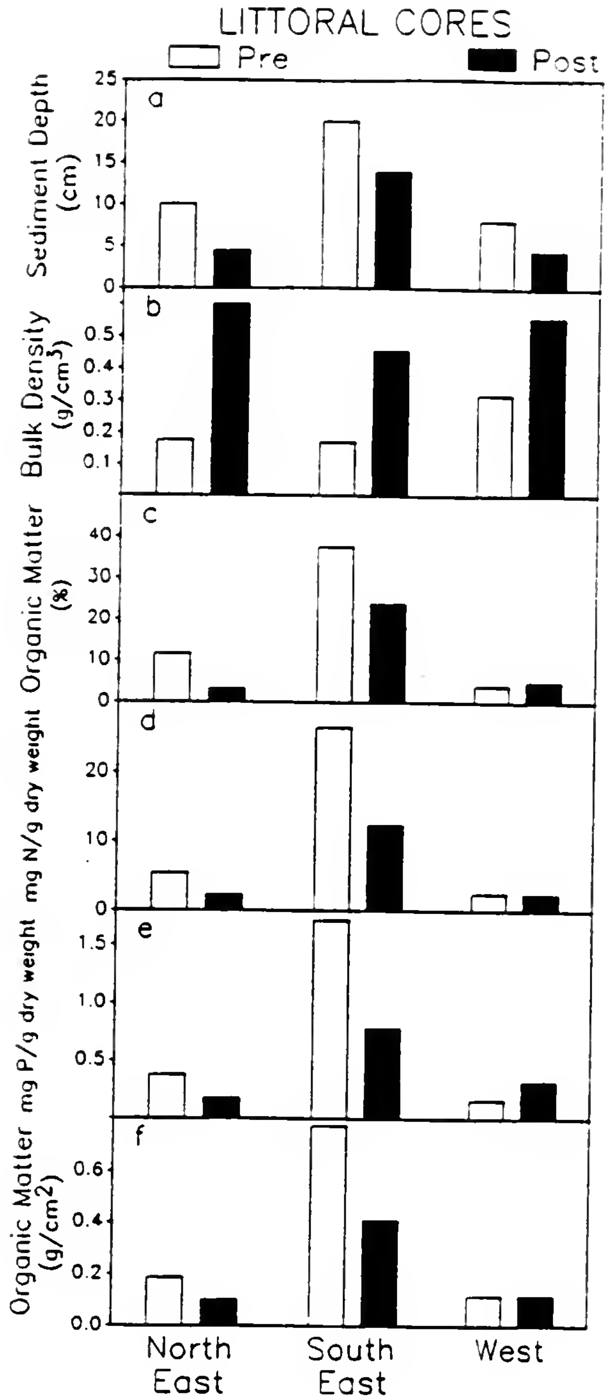


Figure 4-2. Northeast, southeast and west littoral cores, Newnan's Lake. Predrawdown (open bars) and postdrawdown (filled bars); a) sediment depth to sand (cm), b) bulk density (g/cm^3), c) organic matter content (g/g), d) total Kjeldahl nitrogen ($mg_N/g_{dry\ wt}$), e) total phosphorus ($mg_P/g_{dry\ wt}$), and f) organic matter to sand (g/cm^2).

Table 4-2. Material removal rates (g $m^{-2}day^{-1}$) from littoral cores during drawdown, Newnan's Lake.

Removal rates	NE	SE	W
Organic matter	3.52	14.71	0.0
TKN	0.14	1.58	0.01
TP	0.01	0.11	-0.02

Removal rates from the west core deviated from the observed pattern. This may have been due to the location of this station in a rather quiet littoral cove, less subject to wind-wave action. Sediment depth to sand (in cm) decreased at this site following drawdown, but no removal of organic matter (in mg/cm^2) was noted (Figure 4-2f).

These observations may be interpreted in two ways. First, sediment thickness may have decreased without actual removal of material due to consolidation of the substrate at this site. This appears unlikely, since this location remained inundated by at least 30 cm of water at all times. Second, erosion of material did in fact occur, resulting in the observed increased bulk density of the remaining substrate at this site. However, eroded organic material may have been replaced by net primary production during the period of drawdown. Increased organic matter and total phosphorus content at this site (Figure 4-2c, and e) support the latter interpretation. Enhanced primary productivity of benthic algae is likely at low water depth during the drawdown when more light reaches the littoral bottom. This is particularly plausible at the protected west core site, with lower light inhibition by suspended solids.

Removal of littoral sediment likely resulted from increased shear stress caused by wind-wave action. Such surface waves generate periodic oscillations in the water column, which attenuate with water depth (cf. Wetzel 1983) but may reach the sediment-water interface in shallow water. When this stress exceeds the bulk shear strength of the surficial deposits, sediment resuspension occurs (Lick 1982). Waves are largely wind-induced, although wakes from boat traffic may also be considerable in this public-use lake. Resuspended material may then be moved from shallow to deeper parts of the basin by water currents (Davis and Ford 1982; Håkanson 1982; Bengtsson et al. 1990) and eventually settle where water depth is sufficient to eliminate stress from wind-wave action.

The amount of organic matter was not significantly different between the pre- and postdrawdown cores although the northeast and southeast cores demonstrated a considerable reduction (Figure 4-2f). The reduction in thickness of the sediment layer and the increase in bulk density following drawdown, observed at all three sites, were statistically significant ($P=0.05$, paired-t). The assumption in the study was that these three sites were representative of the littoral zone. A different approach may be to make many, simple to perform measurements of sediment-thickness-to-sand throughout the littoral zone pre- and postdrawdown. Such sampling will represent better the entire littoral zone, but it will not provide data on removal rates of bulk sediment, organic matter and nutrients.

Profundal Cores

The locations of the profundal core sites are shown in Figure 4-1. Material in the cores consisted generally of homogeneous, black-brown sediment of low bulk density (<100 mg/cm^3 up to a depth of 30 cm).

Cores 1 and 11. The matching profiles for cores 1 (predrawdown) and 11 (postdrawdown) for bulk density, organic matter, unsupported ^{210}Pb, ^{137}Cs, TKN, and TP are shown in Figure 4-3 (left panel). During sectioning in the laboratory, the first occurrence of clay was noted at 120 cm (core 1) and 132 cm (core 11) (Table 4-3). This suggested an addition of 12 cm of material to the profile at this station during drawdown. Profiles for bulk density and organic matter content revealed a marker at a depth of 82 cm (core 1) and 94 cm (core 11). This also indicated a gain of 12 cm during drawdown.

Unsupported ^{210}Pb and ^{137}Cs profiles, however, implied removal of material postdrawdown. Profiles for TKN and TP were inconclusive with widely fluctuating nutrient levels

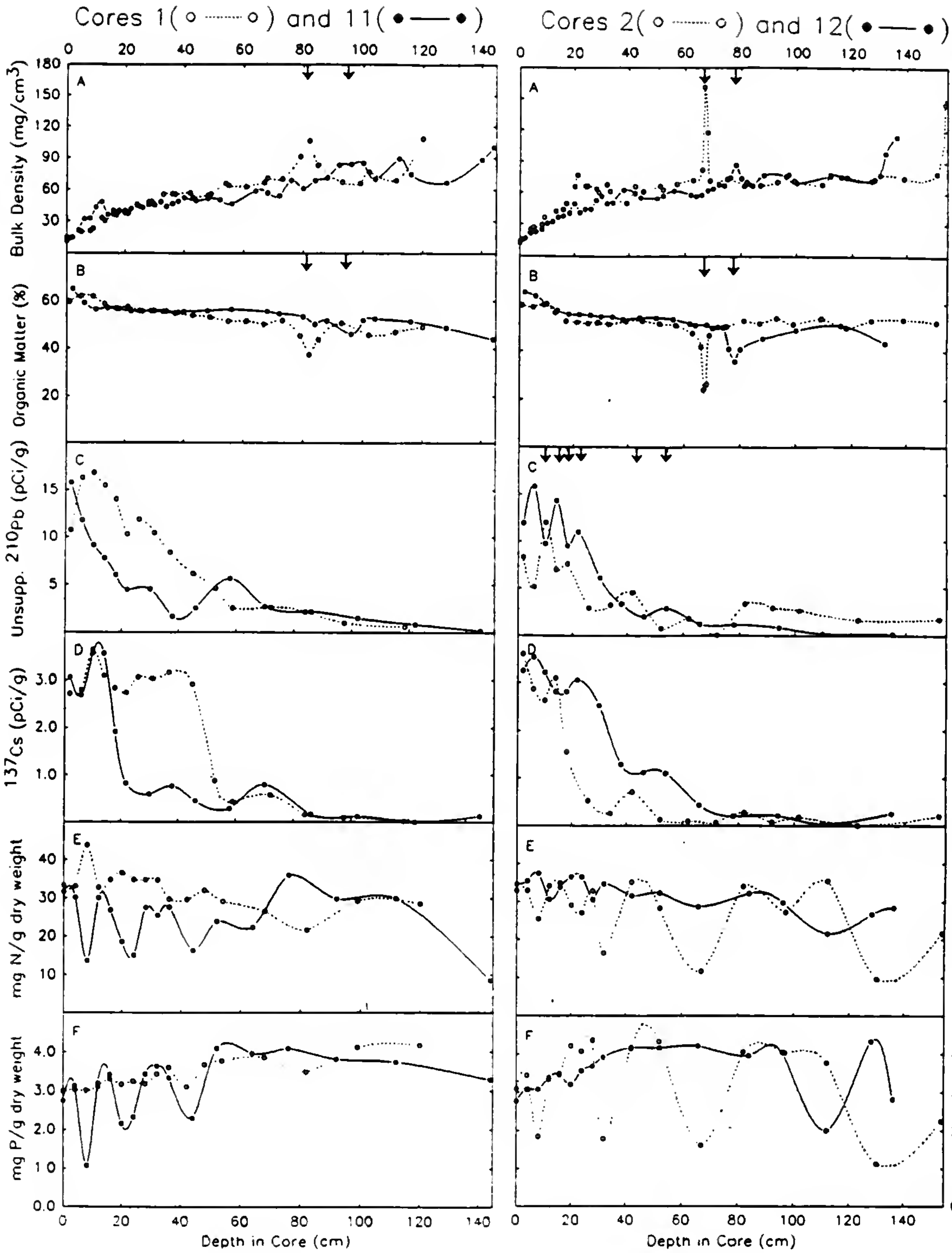


Figure 4-3. Depth profiles for Newnan's Lake cores. Left panel; cores 1 (predrawdown, dashed line) and 11 (post-drawdown, solid line). Right panel; cores 2 (predrawdown, dashed line) and 12 (postdrawdown, solid line). Arrows indicate horizons discussed in the text. A, bulk density in mg/cm^3; B, organic matter content in mg/mg; C, unsupported Pb-210 in pCi/g; D, Cs-137 in pCi/g; E, total Kjeldahl nitrogen in $mg/g_{dry\ wt}$; and F, total phosphorus in $mg/g_{dry\ wt}$.

in both cores. Average TKN concentrations in the top 120 cm of cores 1 and 11 were 33 and 27 $mg_{TKN}/g_{dry\ weight}$ respectively. TP levels averaged 3.4 and 3.0 $mg_{TP}/g_{dry\ weight}$. Examination of the six profiles did not demonstrate a distinct removal or gain of bottom material due to drawdown for this station.

Table 4-3. Length of retrieved cores and depth of sand and clay horizons in Newnan's Lake cores.

Core site	Core length (cm)	Depth (cm) to sand(s)/clay(c)
1	129	120c
11	148	132c
2	155	
12	140	
5	113	85s,113s,95c
15	92	76s,86c
6	135	135c
16	115	
7	135	100s
17	148	100s
8	72	38s,64s
18	61	36s

Cores 2 and 12. At station 2, bulk density and organic matter profiles (Figure 4-3, right panel) revealed clear markers at 68 cm predrawdown (core 2) and 78 cm postdrawdown (core 12). Peaks and dips in the unsupported ^{210}Pb and ^{137}Cs profiles displayed a similar gain of approximately 10 cm of material in the postdrawdown core.

Predrawdown peaks in the ^{210}Pb profile at depths of 10, 18, and 44 cm occurred in the postdrawdown core at 16, 24, and 54 cm. A similar gain was evident from the ^{137}Cs profiles. This indicated an addition of approximately 9 cm during drawdown at this station.

Based on the bulk density of the top 10 cm layer of the postdrawdown core, this gain was equivalent to 0.18 g/cm^2. This translated into an additional 1.1 years of sediment deposition during the 245-day drawdown period compared to normal, non-drawdown sedimentation rates (discussed later). TKN and TP profiles were inconclusive. TKN levels fluctuated between 25 and 35 $mg_{TKN}/g_{dry\ weight}$ with a few measurements below 20 $mg_{TKN}/g_{dry\ weight}$. Averages for both cores approximated 32 $mg_{TKN}/g_{dry\ weight}$ and corresponded to levels in cores 1 and 11. Fluctuations in TP were even more pronounced with an average concentration between 3.0 and 3.5 $mg_{TP}/g_{dry\ weight}$.

Cores 5 and 15. Visual analysis of cores 5 (predrawdown) and 15 (postdrawdown) during sectioning revealed distinct sand layers at depths of 85 and 113 cm for core 5 and at 76 cm for core 15. First occurrence of clay was at 95 cm (core 5) and 86 cm (core 15) (Table 4-3). Based on these marker horizons, 9 cm of sediments were removed during drawdown. Bulk density and organic matter profiles appeared to display a phase difference of a minimum of 10 cm (Figure 4-4, left panel) also showing

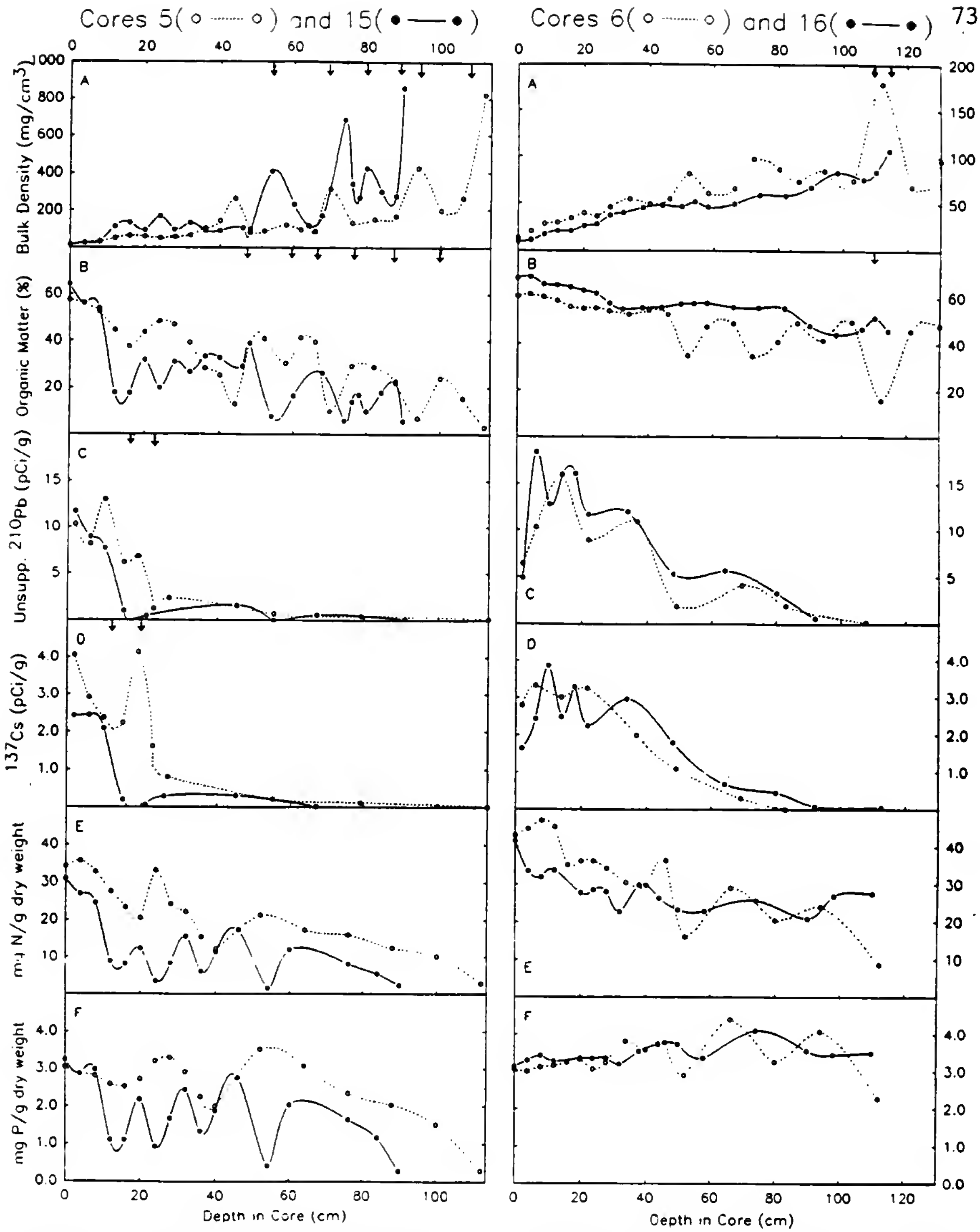


Figure 4-4. Depth profiles for Newnan's Lake cores. Left panel; cores 5 (pre-drawdown, dashed line) and 15 (post-drawdown, solid line). Right panel; cores 6 (pre-drawdown, dashed line) and 16 (post-drawdown, solid line). Arrows indicate horizons discussed in the text. A, bulk density in mg/cm^3; B, organic matter content in mg/mg; C, unsupported Pb-210 in pCi/g; D, Cs-137 in pCi/g; E, total Kjeldahl nitrogen in $mg/g_{dry\ wt}$; and F, total phosphorus in $mg/g_{dry\ wt}$.

material removal during drawdown. Unsupported ^{210}Pb and ^{137}Cs profiles indicated a distinct removal of approximately 8 cm during drawdown. In addition, TKN and TP profiles suggested material removal postdrawdown. This implied that the drawdown removed a minimum of 8 cm, or 0.26 g/m^2, from the sediment profile. This equated to approximately 4.7 years of sediment removal at this site during drawdown (discussed later).

The profiles also demonstrated the different character of the substrate at station 5-15 compared with other sampling sites. Bulk densities were significantly higher throughout the profiles, while organic matter and nutrient concentrations were depressed compared with deposits at similar depths in other locations. Field observations concurred with these data in that substrate appeared firmer, with a relatively high sand content.

<u>Cores 6 and 16</u>. Sectioning of the cores in the laboratory revealed a clay horizon in core 6 (predrawdown) at 135 cm (Table 4-3), well below the bottom section of core 16 (postdrawdown). Bulk density and organic matter profiles (Figure 4-4, right panel) showed a rather homogeneous top 100 cm for both cores containing flocculent material (bulk density<100 mg/cm^3) and an average organic matter content of 60% (weight/weight ratio). Firmer substrate started at 105 cm depth in core 6 and at approximately 110 cm in core 16. A horizon of inorganic

material was encountered first at 105 cm depth in core 6. This layer may have occurred immediately below the last sampled segment in core 16.

Isotope profiles for both cores also displayed a shift in phase of several centimeters, while nutrient profiles appeared inconclusive. Average TKN and TP levels coincided with cores 1-11 and 2-12 (30 $mg_{TKN}/g_{dry\ weight}$ and 3.0-3.5 $mg_{TP}/g_{dry\ weight}$). In summary, these profiles provided weak support for the conclusion that 5 cm, or 0.05 g/cm^2, of material was deposited at this site during drawdown. This equated to approximately 0.04 years of sedimentation in excess of "normal" sedimentation occurring during the period of drawdown (discussed later).

Cores 7 and 17. Core sectioning in the laboratory demonstrated 1 m of black, soft, organic muck overlying a distinct firm, sandy layer in both cores 7 (predrawdown) and 17 (postdrawdown) (Table 4-3). Water and organic matter content increased again below this layer. This was reflected in the bulk density and organic matter profiles at this core site (Figure 4-5, left panel).

The additional similarity of unsupported ^{210}Pb and nutrient profiles for pre- and postdrawdown cores suggested no significant redistribution of material at this site. The absence of a ^{137}Cs peak below 40 cm in the postdrawdown core is unexplained, but has been verified by repeated analysis of the sample material. Average TKN and TP

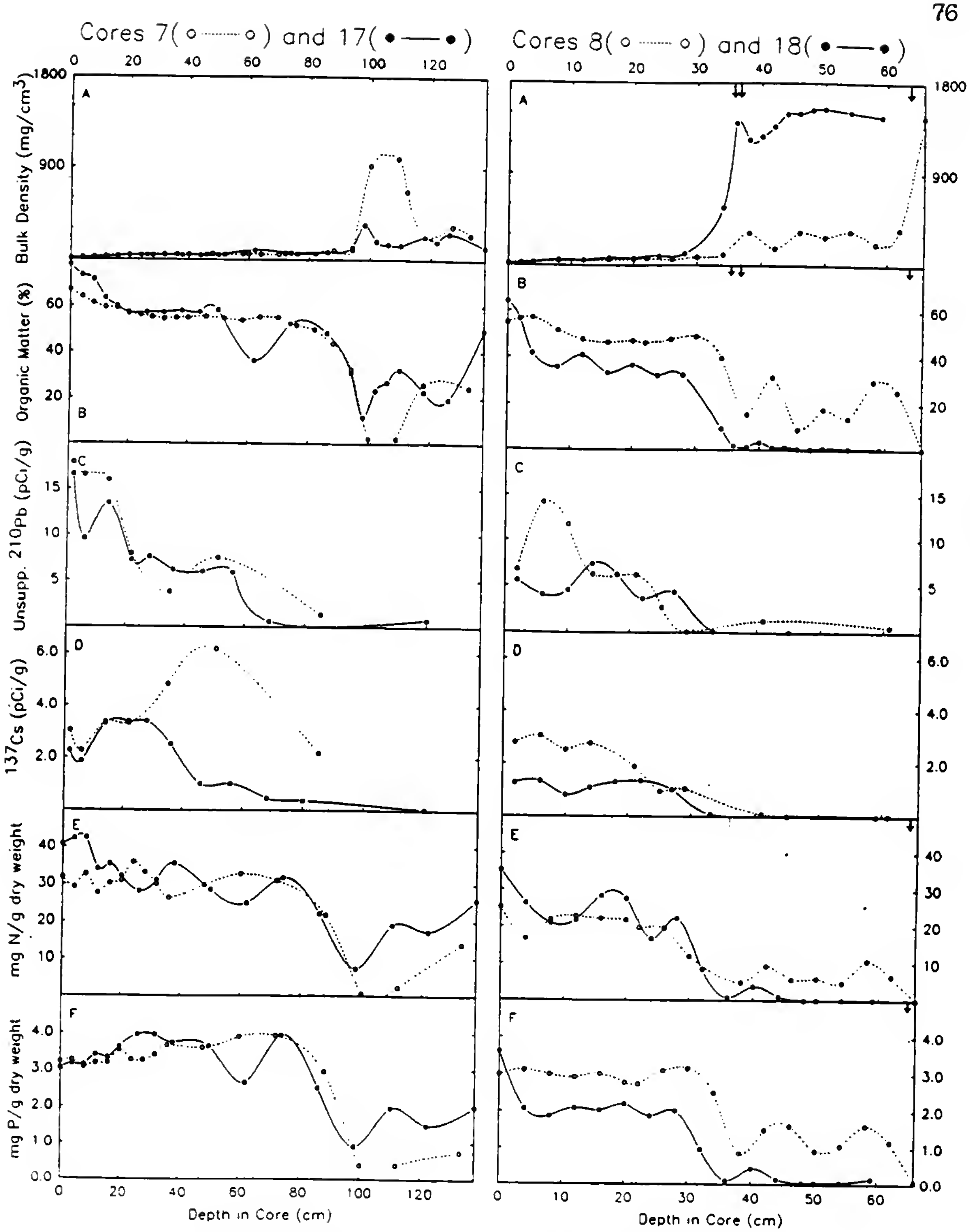


Figure 4-5. Depth profiles for Newnan's Lake cores. Left panel; cores 7 (pre-drawdown, dashed line) and 17 (post-drawdown, solid line). Right panel; cores 8 (pre-drawdown, dashed line) and 18 (post-drawdown, solid line). Arrows indicate horizons discussed in the text. A, bulk density in mg/cm^3; B, organic matter content in mg/mg; C, unsupported Pb-210 in pCi/g; D, Cs-137 in pCi/g; E, total Kjeldahl nitrogen in mg/g$_{dry\ wt}$; and F, total phosphorus in mg/g$_{dry\ wt}$.

concentrations in the top 100 cm corresponded to levels found in other cores (except site 5-15).

Cores 8 and 18. Minor sand was encountered at 38 cm in core 8 (predrawdown), with a clear, pronounced sand layer at 64 cm. Core 18 (postdrawdown) showed clear sand at 36 cm (Table 4-3). Both the bulk density and organic matter profiles displayed these markers (Figure 4-5, right panel) with bulk density values in excess of 1200 mg/cm^3 and organic matter content less than 5%. Likewise, TKN and TP concentrations approached zero at 65 (core 8) and 36 cm (core 18). Evidence from these four profiles suggested that approximately 28 cm of material eroded from the profile at this station during drawdown.

The absence of the ^{210}Pb peak in the top 10 cm of the postdrawdown core also suggested such removal, but to a much lesser extent. Analogously, elimination of the ^{137}Cs peak in the top 20 cm of deposits in core 18 and the onset of detectable levels of this radionuclide about 10 cm deeper in the predrawdown profile implied removal of material during drawdown. Such massive removal rates, however, seemed implausible in light of the small flushing rates of particulate matter through the nearby outflow recorded during drawdown (Gottgens and Crisman 1991). It is more likely that either the exact location of the pre- and postdrawdown cores did not match or that a significant disturbance of the sediment profile occurred at this station during drawdown. Hence, no inference on gain or removal of bottom material at this site was made.

All stations

Rates of removal or gain of material for the profundal sampling stations are summarized in Table 4-4. The 245-day time period covered started immediately

Table 4-4. Profundal cores: Gain/removal (-) rates of bulk sediment, organic matter, TKN, and TP (g $m^{-2}day^{-1}$).

Gain/removal(-) rate	Core station 1-11	2-12	5-15	6-16	7-17	8-18
Bulk sediment	*	7.2	-10.6	2.2	0.0	*
Organic matter	*	4.3	-5.9	1.5	0.0	*
TKN	*	0.2	-0.3	0.1	0.0	*
TP	*	0.02	-0.03	0.01	0.0	*

*) No clear record or conflicting evidence.

prior to removal of the spillway and lasted until the natural return of the water level to predrawdown levels following spillway re-installation.

Consequently, out of six profundal cores analyzed for this study, two showed evidence of added sediment deposition during drawdown, one showed no gain or loss, one showed removal, and two were inconclusive. No conclusion, therefore, can be reached by this study's data regarding flocculent sediment accumulation in the profundal zone of Newnan's Lake during the drawdown.

While sediment removal from the littoral zone was indicated by the results of this study, a quantitative record of this transfer to profundal sites was difficult to obtain. This is particularly true when profundal substrate consisted of a thick pack of near homogeneous material without clear marker horizons. Furthermore, the small drop in lake level during this drawdown did not create large areas of erosion in the littoral zone. This reduced the magnitude of potential redistribution of sediments in the lake which reduced the signal in profundal core profiles.

Using the ^{210}Pb profiles, sedimentation rates were computed for each profundal station. Calculations followed the constant rate of supply model (Goldberg 1963; Appleby and Oldfield 1983). These accumulation rates were then compared to the gain or loss of material at those sites due to drawdown. As such, this gain or loss was equated to a time period of "normal" sedimentation. For instance, the gain of material due to drawdown at station 2-12 was equivalent to 1.1 years of sedimentation (Table 4-5).

Uncertainty analysis

Because inferences were made from ^{210}Pb profiles in different aspects of this work, an assessment of the level of confidence in these profiles is appropriate. This was particularly pertinent since profiles were established in soft lake sediment with the potential of disturbance in the chronology of deposits. Three independent observations aided in such an assessment.

Table 4-5. Newnan's Lake profundal cores: Comparisons of recent (5 yrs. B.P.) dry-sediment accumulation rates with gain/loss (-) of material during drawdown, and with calculated dry-sediment accumulation rates.

Core site	Recent sed.rt. ($g\ cm^{-2}yr^{-1}$)	Gain/loss during drawdown (g/cm^2)	Gain/loss during drawdown (yrs) [1]	Calc.recent sed.rt. [2] ($g\ cm^{-2}yr^{-1}$)	Cumulative residual uns. ^{210}Pb (pCi/cm^2)
1-11	0.06	*	*	0.05	30.2
2-12	0.10	0.18	1.10	0.10	26.4
5-15	0.07	-0.26	-4.70	0.07	17.4
6-16	0.08	0.05	0.04	0.07	28.0
7-17	0.06	0.00	-0.70	0.05	27.0
8-18	0.07	*	*	0.07	18.9

*) no clear record or conflicting evidence.
1) sedimentation normally occurring during the 245 days between pre- and post-measurements is subtracted.
2) Using a model developed by Binford and Brenner (1986). Measured average cumulative residual unsupported ^{210}Pb=24.65 pCi/cm^2; Flux for ^{210}Pb-fallout=0.77 $pCi\ cm^{-2}yr^{-1}$.

First, recent ^{210}Pb based deposition rates were not statistically different (P=0.05; two-tailed correlated test) from calculated values using an earlier, independently developed model (Binford and Brenner 1986) (Table 4-5). In this model fallout ^{210}Pb is used as a dilution tracer to compute net accumulation rates of any material in surface mud according to:

$$r= F_{210Pb} \times A^{-1} \qquad (4\text{-}1)$$

where F_{210Pb} equals the flux of fallout ^{210}Pb ($pCi\ cm^{-2}y^{-1}$) and A is the activity of ^{210}Pb in the sediment sample

($pCi/g_{dry\ weight}$). Furthermore, the different cores from Newnan's Lake had comparable ^{210}Pb residuals (i.e. total residual unsupported ^{210}Pb contents) despite differences in accumulation rates (Table 4-5). The ^{210}Pb residuals of the cores reflected the ^{210}Pb fallout from the atmosphere. Since this fallout lies in the range 0.5-0.9 pCi $cm^{-2}yr^{-1}$ (Nozaki et al. 1978), depending on locality, the ^{210}Pb residuals should lie in the range 16-30 pCi/cm^2 (the ^{210}Pb radioactive decay constant=0.03114 yr^{-1}). This corresponded to the measurements in Newnan's Lake cores (Table 4-5).

Second, recent ^{210}Pb based dry-sedimentation rates correlated well with water depth (R^2=0.81; N=5), when station 5-15 is excluded. The proximity of this site to a fish attractor (e.g. submersed brush attached to an anchored buoy) and, hence, higher boat traffic and boat wake may have disturbed the sediments and depressed sedimentation rates. Consequently, direct comparisons between this sampling site and others in the lake may be misleading. The close relationship between water depth and material accumulation rates is well-established (Evans and Rigler 1980). Including the 5-15 site, 57% of the variability in dry-sedimentation rates was explained by depth of the water column. While this relationship does not necessarily underwrite the accuracy of recent ^{210}Pb levels, it does demonstrate that these levels correlate well with each other (i.e. their precision is supported).

Finally, counting statistics and error prediction were applied to the recorded unsupported ^{210}Pb data to compute statistical precision. This error analysis only addressed internal uncertainty associated with the accuracy of γ-ray detection. It did not consider uncertainty controlled by external factors such as error associated with the (sub) sampling design, smearing of the core (Chant and Cornett 1991), bulk density determinations, and post depositional mobility of constituents in the core (Anderson et al. 1987). "Error bars" associated with the experimental data are illustrated for cores 1 and 2 in Figure 4-6. Other cores showed a similar magnitude of error. The length of the error bar equals one standard deviation (σ) on either side of the point (i.e. 68.3% confidence limits), which is standard practice in expressing uncertainty in nuclear measurements (Wang et al. 1975). Because the recorded counts in nuclear counting experiments follow a Poisson distribution (Knoll 1979), the predicted standard deviation is the square root of the mean number of counts. To arrive at this mean (counts/hr), samples were counted from 14 to 45 hours depending on sample weight. Since

$$\text{Net counts} = \text{Total counts} - \text{Background counts} \qquad (4\text{-}2)$$

uncertainty in the net counts is propagated according to

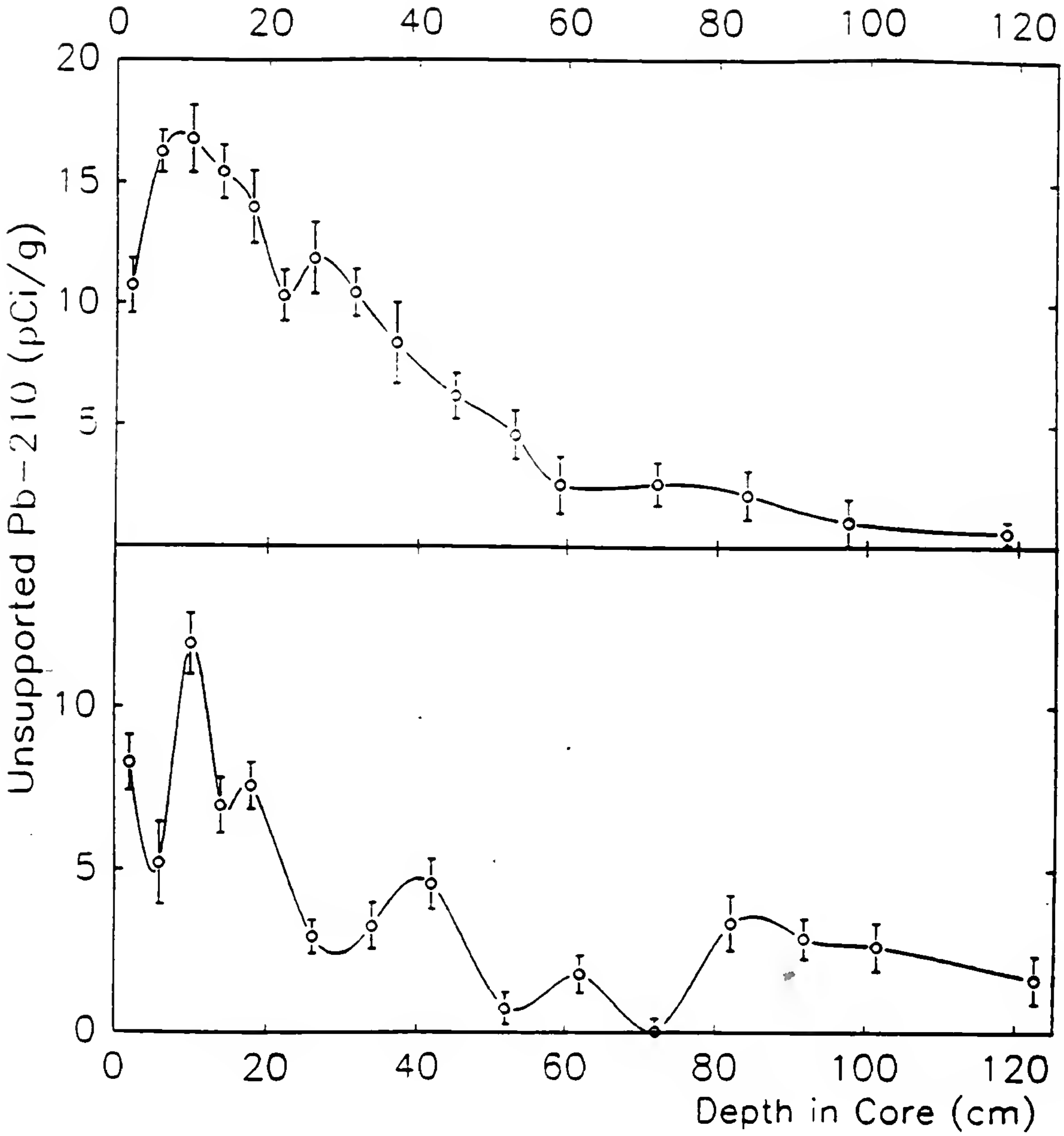


Figure 4-6. Unsupported Pb-210 profiles for core 1 (top) and core 2 (bottom). The length of the error bar equals 1σ on either side of the data point.

$$\sigma_n = \sqrt{(\sigma_t^2 + \sigma_b^2)} \quad (4\text{-}3)$$

where σ_n, σ_t, and σ_b are, respectively, the standard deviations of the net count, total count, and background count.

The effect of the computed internal uncertainties on the interpretation of the profile in terms of marker horizons or sediment accumulation rates was small. For instance, recent sediment accumulation rates for core 1 varied between 0.049 and 0.055 g $cm^{-2}yr^{-1}$ when mean unsupported ^{210}Pb concentrations (5 years B.P.) plus and minus 1 σ, respectively, were substituted into equation (4-1). Analogously, these rates for core 2 ranged from 0.085-0.105 g $cm^{-2}yr^{-1}$. Error analysis, such as Monte Carlo simulation, may be used to estimate the effect of internal uncertainty on the assigned dates for an entire core profile. Binford (1990) applied this technique to 12 cores from north Florida lakes and found 95% confidence intervals ranging from about 1-2 years at sediments 10 years of age, 10-20 at 100 years, and 8-90 at 150 year old deposits. No estimates of the effect of external uncertainty on ^{210}Pb derived sediment chronology have been documented.

Summary and Conclusions

An average of 6.08 g $m^{-2}day^{-1}$ of organic matter (0.58 g_{TKN} $m^{-2}day^{-1}$ and 0.03 g_{TP} $m^{-2}day^{-1}$) eroded from the littoral zone by lowering the water depth from 62 to 30 cm for 8 weeks. Littoral sediments with low bulk density eroded fastest, and bulk density of remaining substrate increased by an average of 250%. In the absence of direct field evidence of distinct stratigraphy, transfer of this material to deeper areas of the lake was measured using sedimentary cores with marker horizons. The latter were derived from measurements of bulk density, organic matter, radionuclides, and nutrients. The quantitative record of this transfer, however, was unclear. The small drop in lake level during drawdown may have contributed to this. Since the drawdown did not create large areas of erosion in the littoral zone, the magnitude of potential redistribution of sediments in the lake and, thereby, the signal in profundal cores, is reduced.

Two options may now be identified for further work. First, measurements limited to the littoral zone can be followed by calculations of deposition rates in the profundal if the extent of the zones of erosion, transport, and accumulation of sediment in the lake are known. Second, use of settling sediment traps to collect resuspended and transported material may give additional insight. In Newnan's Lake, however, installation of these traps on the profundal bottom is difficult because of the soft nature of the substrate. Suspension of the traps in the water column may jeopardize public use of the lake.

Instead, this technique may be used in other systems. Since resuspension and erosion of fine sediments may influence the ecology of a lake through habitat alteration, release of nutrients, high turbidity and enhanced sediment oxygen demand, erosion processes must be considered when drawdowns are attempted.

CHAPTER 5
SEDIMENT AND PHOSPHORUS DYNAMICS MODEL

Analysis of field data demonstrated that the spillway caused accelerated accumulation of flocculent and nutrient-rich sediment in Newnan's Lake (Chapter 2). Plausible consequences of such increased material transfer between water and sediment for shallow, productive lake ecosystems were discussed in the introduction (Chapter 1). They included: (1) increased resuspension of flocculent sediments by wind-wave action leading to high turbidity and diminished light penetration in the water column, (2) enhanced biological and chemical oxygen demand from organic and inorganic suspended matter (Hargrave 1969; James 1974), and (3) expansion of flocculent substrate into the littoral zone.

Each of these consequences, in turn, has implications for lake management criteria such as water clarity, level of primary production, extent of the littoral zone, lake access, and standing stock of sportfish. When acting simultaneously, however, their net effect on these lake management criteria is difficult to determine. A few examples may illustrate this. Increased suspended matter in the water column, for instance, decreases light penetration and habitat for submerged vegetation, but accelerated deposition of sediments and the resulting increased rate of filling of the lake may actually lead to a larger area that can be invaded by bottom vegetation.

Increased suspended matter may also reduce water clarity and decrease the aesthetic value of the lake. This opposes management objectives designed to secure the lake for water recreation. Enhanced primary production may follow when higher rates of wind-wave induced resuspension of bottom material increase internal loading of nutrients in the lake through entrainment of nutrient-rich interstitial (pore) water from the surface sediments (Lam and Jaquet 1976) or desorption from suspended particles (Pollman 1983). Oxygen demand from increased suspended solids lowers dissolved oxygen level in the water column, which may promote nutrient release from the sediments (Mortimer 1971; Theis and McCabe 1978).

Low concentrations of dissolved oxygen may stress fish and other heterotrophs and selectively favor species with physiological and behavioral mechanisms that aid them in surviving under such conditions. The Florida gar (McCormack 1967), bowfin (Johansen 1970), and bullheads (Loftus and Kushlan 1987) are able to utilize atmospheric oxygen, while small fishes with upturned mouths can extract oxygen from the well-oxygenated surface film (Lewis 1970).

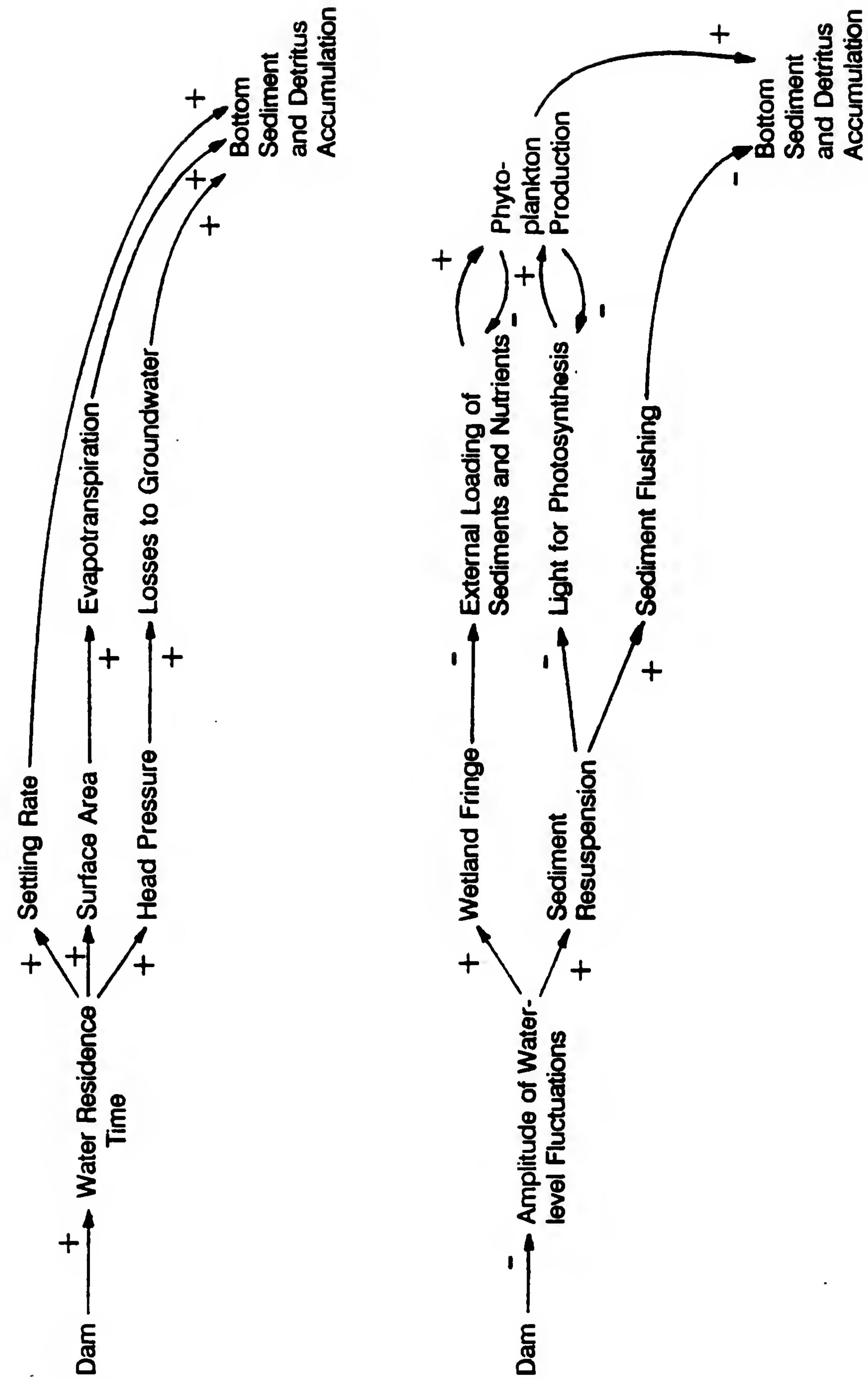
Centrarchids, a significant component of the standing stock of sportfish in Florida lakes, are usually the first fishes to die under low-oxygen stress (Kushlan 1974). This would conflict with lake management strategies aimed at promoting sportfish. Expansion of flocculent sediments into the littoral zone may also reduce nesting habitat for centrarchids (Bruno 1984), which rely on firm substrate for deposition of eggs (Eddy and Underhill 1978).

Changes in components of the Newnan's Lake ecosystem have been documented, such as the consistent decline in the number of harvestable sportfish per hectare from 1850 to 227 between 1982 and 1988 (Florida Game and Fresh Water Fish Commission 1982-1989). However, no work has been done to analyze the response of the lake to simultaneously acting factors and assumptions such as those described above. In other systems, dynamic simulation models have been used successfully for such analysis by synthesizing the best current understanding of succession in a prairie ecosystem (Gutierrez and Fey 1975), eutrophication in a lake (Anderson 1973), and predator-prey dynamics in a natural two-species system (Montague et al. 1982). The computer model keeps track of all interrelationships in the system once they have been identified. It then builds on the reliable part of our understanding of the system, while compensating for the "unreliable" part by requiring an explicit statement of assumptions and showing the consequences of these in a model analysis. Hence, the first objective of this model is to integrate existing information concerning the Newnan's Lake ecosystem and to program this information on the computer in the form of a testable hypothesis.

A second purpose of the model is to identify knowledge gaps that are most critical in improving the understanding of the behavior of this ecosystem. This is accomplished by systematically changing the value of the parameters used to develop the model and determining the effect of this systematic change on model output. Those parameters generating considerable changes in one or more model output characteristics require further field and/or laboratory study to increase confidence in the model. As such, the model helps to guide limnological research, while, in turn, research data can be used to further develop the model.

Finally, the model aims to evaluate the consequences of alternative management actions in Newnan's Lake. Models of different hypotheses of the causes of past lake responses (such as an increase in the rate of sediment deposition) to perturbation (such as water-level stabilization) may be used to make inferences about changes to be expected following future lake management actions. Different hypotheses by which the dam may influence sediment and detritus accumulation are illustrated in Figure 5-1. They include spillway-induced changes in lake

Figure 5-1. Hypotheses by which the dam may influence sediment and detritus accumulation rates in Newnan's Lake, Florida. Causality in the hypothesized relationships is implied by showing the influencing parameter at the arrow tails and the influenced parameter at the arrow head. Positive and negative signs at the arrow heads indicate whether the parameter at the tail of the arrow and the parameter at the head of the arrow change in the same or in the opposite direction of the influencing parameter respectively.


Dam
Water Residence Time
Settling Rate
Surface Area
Head Pressure
Evapotranspiration
Losses to Groundwater
Bottom Sediment and Detritus Accumulation
Dam
Amplitude of Water-level Fluctuations
Wetland Fringe
Sediment Resuspension
External Loading of Sediments and Nutrients
Light for Photosynthesis
Sediment Flushing
Phyto-plankton Production
Bottom Sediment and Detritus Accumulation

hydrology (e.g. water residence time), in sediment physics (e.g. sediment resuspension and flushing), and in primary production (e.g. light and nutrient conditions). By testing the effect of each of these mechanisms on model output and ranking them in terms of risks and benefits to intended uses of the lake, the model can provide options for lake management. As an example, the possible effects of installation of a dam designed to release bottom-water, i.e. one which promotes sediment flushing, can be tested in the model. Similarly, the effects of hand-planting of native aquatic macrophytes, a recently started program (Krummrich, Florida Game and Fresh Water Fish Commission, pers. comm.), or a proposed pumpdown of the lake (KBN 1991) may be predicted.

Methods

Feedback Dynamics

Different dynamic modeling methods (as compared with static modeling of flow analysis, energy analysis, or statistical modeling) may be used to develop and test causal hypotheses for ecosystem behavior. Two of these methods are the energy modeling techniques of Odum (1983 1988) and feedback dynamics, originally described by Forrester (1961, 1968). Odum's modeling approach, based on energy flow in open systems, has been used to test hypothetical scenarios relative to lake management (Gayle 1975; Fontaine 1978). Feedback dynamics emphasizes the identification of a closed system of influences (i.e. a system of feedback loops) that produces time patterns in an ecosystem. By limiting the analysis to an endogenous, closed system of feedback loops, attention is focused on those influences that are most important in generating and controlling system behavior. External features, such as variables that only vary as a function of time (e.g. seasons) and random variation are excluded. They do not require an explanation and can not be managed to generate different system behavior. Feedback dynamics methodology was used in the development of this simulation model.

The immediate goal is to develop a "dynamic hypothesis", which explains the observed pattern of changes in the ecosystem. The feedback relationships involved in the hypothesis are quantified using field and laboratory data, literature values, theory and opinion. Next, this hypothesis is represented in equation form and the solution is simulated on the computer. This solution is then analyzed to determine whether the model is consistent with theory and a plausible representation of what is known to be true about the ecosystem of interest. If the model fails this test, the hypothesis is rejected and a modified hypothesis may be formulated and tested. In addition, a sensitivity analysis is done in which the consequences of inaccurate parameter estimates are determined. Parameters that produce substantial changes in one or more output characteristics when altered from their initial values are key areas for further field and laboratory research to enhance confidence in the model. Finally, model validation (i.e. a rigorous attempt to falsify the dynamic hypothesis) is carried out. This may involve field testing of the model, such as comparing the results of a change in the model with the outcome of the same change made in the field. A flow chart of the feedback dynamics procedure is given in Figure 5-2. The coupling between limnology and modeling is emphasized in that field and laboratory research assist in model development and analysis (particularly in recording initial observations, quantifying the feedback relationships, and field testing the model), while the model analysis identifies the most sensitive areas for further research. As such, the model is a useful tool in the analysis of complex ecosystems.

Dynamic Hypothesis

The overall hypothesis is that the spillway in the outlet of Newnan's Lake influences a system of feedback loops in the lake due to increased water residence time, and reductions in both bottom-water drainage and amplitude of seasonal water-level fluctuations. These influences contribute to the lake management problems described in the introduction of this chapter. Feedback relationships involved in the modeled hypothesis for the profundal zone are illustrated in Figure 5-3. Causality in the

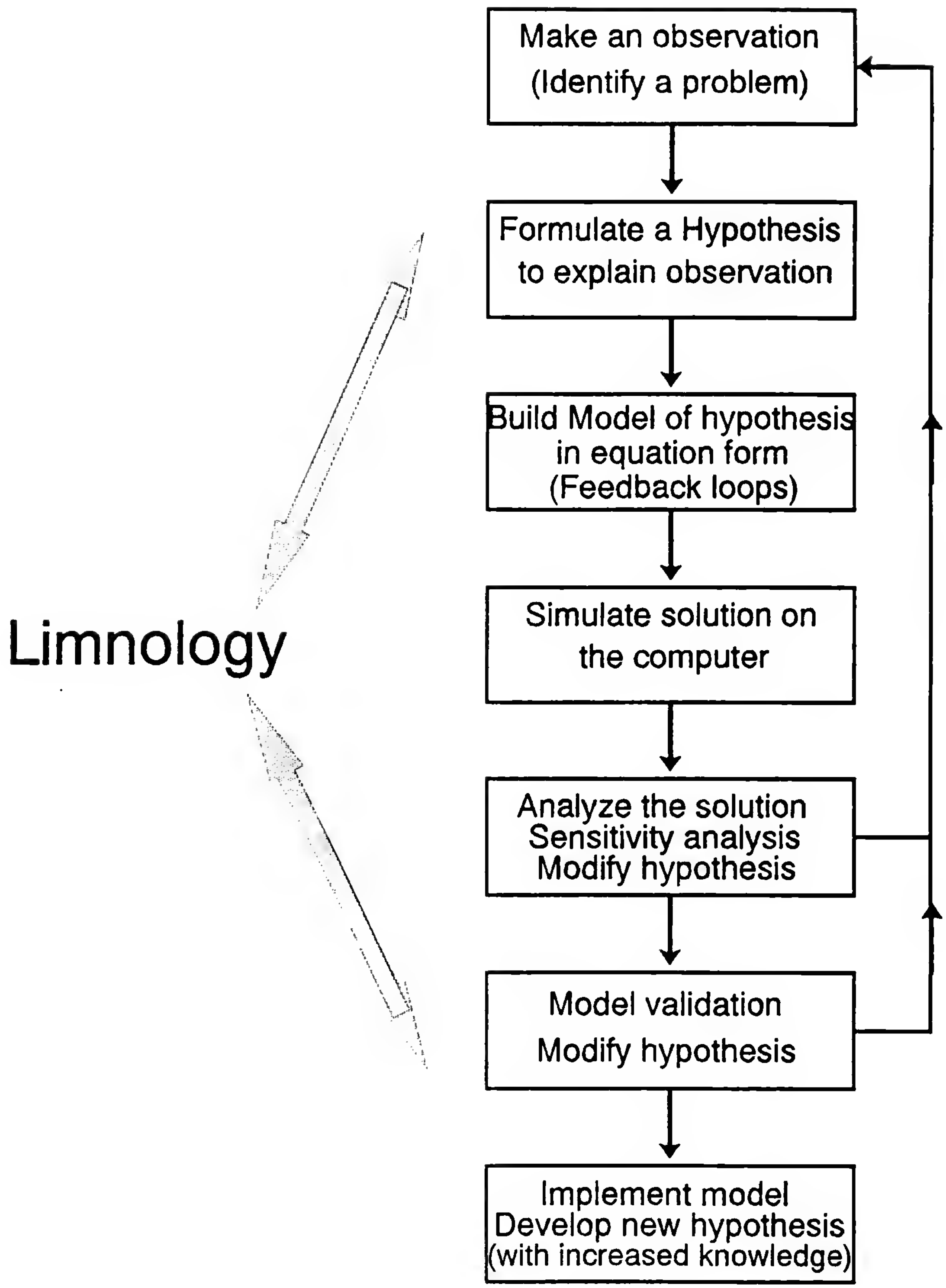
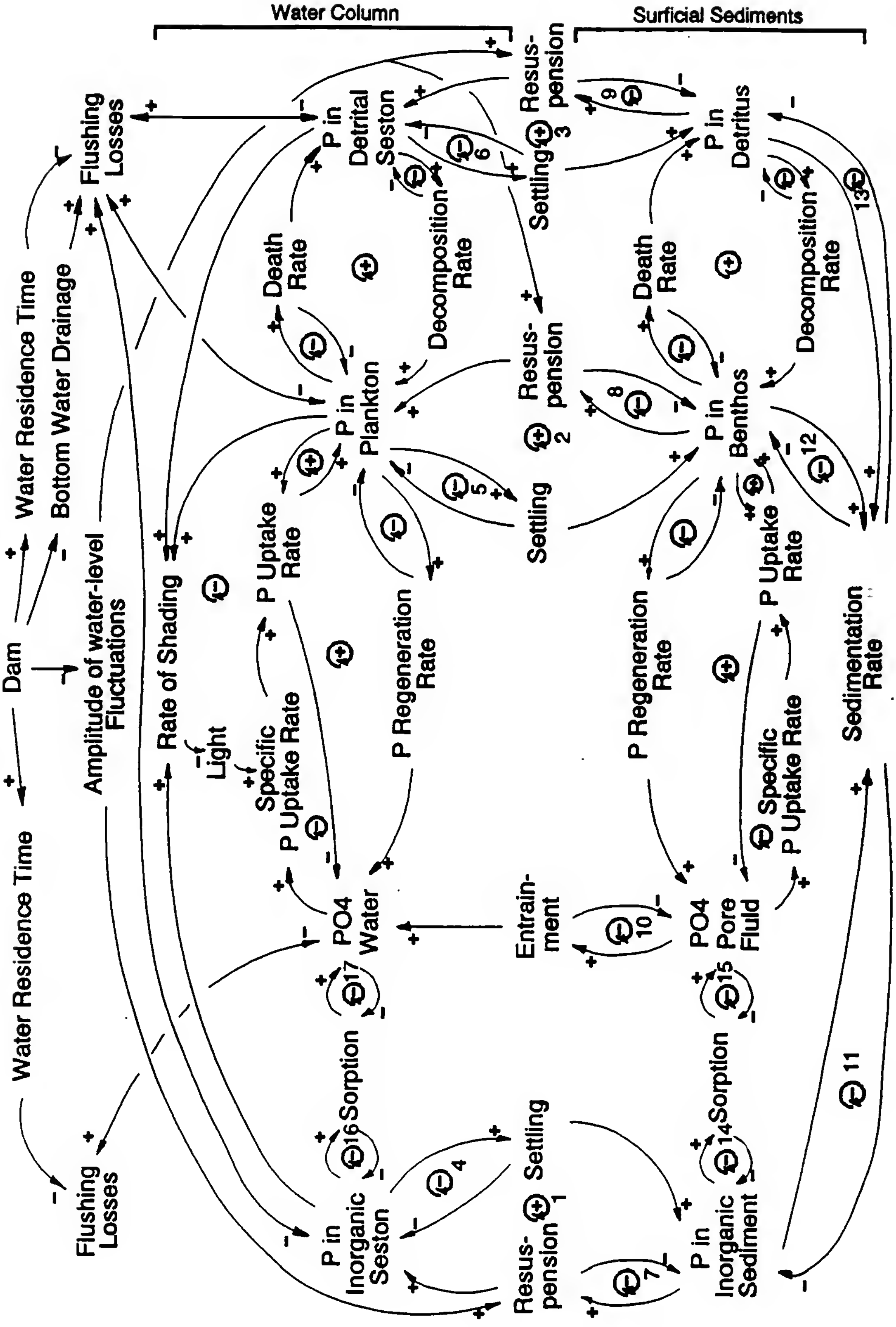


Figure 5-2. A flow-chart of the feedback dynamics procedure and its interaction with limnological research.

Figure 5-3. Influence diagram of the modeled hypothesis for the profundal zone. See text for feedback-loop descriptions.


Water Column
Surficial Sediments
Dam
Water Residence Time
Bottom Water Drainage
Amplitude of water-level Fluctuations
Flushing Losses
Rate of Shading
Light
Specific P Uptake Rate
P Uptake Rate
P Regeneration Rate
P in Detrital Seston
P in Plankton
P in Inorganic Seston
PO4 Water
Death Rate
Decomposition Rate
Sorption
Resus-pension
Settling
Entrain-ment
P in Detritus
P in Benthos
P in Inorganic Sediment
PO4 Pore Fluid
Sedimentation Rate

hypothesized relationships is implied by showing the influencing parameter at the arrow tails and the influenced parameter at the arrow heads. Positive and negative signs at the arrow heads indicate whether the parameter at the tail of the arrow and the parameter at the head of the arrow change in the same or in the opposite direction, respectively. The overall sign of a closed loop is indicated by a sign within a small circular arrow within the loop. In a positive feedback loop, the original direction of any change in any loop component is reinforced. Conversely, a negative loop opposes such change. The parameters used in the simulation model are summarized in Table 5-1. The simulation model listing is given in Appendix B. Quick Basic 4.5 (Microsoft Corp.) was the programming language used to develop the computer code.

Profundal. Causal feedback relationships hypothesized for the profundal zone of the lake (Figure 5-3) consist of water-column and surface-sediment components with processes of resuspension and settling occurring at the sediment-water interface. The importance of phosphorus (P) in controlling primary production in this lake was documented in Chapter 1. Therefore, P is used as a tracer of material in this model, and components are represented in grams of P per m^2 (in the surface sediment) or grams of P per m^3 (in the water column). Similarly, resuspension and settling are computed as grams of P per m^2 per year or grams of P

Table 5-1. Model parameter values. See text for rationale of estimated values.

Constant	Value	Units	Description	Source of value
AMPNODAM	0.55	m	Amplitude of water-level fluctuations (no dam)	Skoglund 1990; Adkins 1991
AMPDAM	0.39	m	Amplitude of water-level fluctuation (with dam)	Ibid.
RISEDAM	0.13	m (msl)	Rise in water-level following dam installation	Ibid.
FREQLEVEL	50		Frequency of water-level fluctuations	Operationally defined
RAMPDURATION	0.5	y	Ramp coefficient for water-level rise	Estimate
BWDNODAM	0.5	$g\ m^{-3}$	Seston concentration in outflow (no dam)	Field measurement
BWDDAM	0.3	$g\ m^{-3}$	Seston concentration in outflow (with dam)	Field measurement
HYDRESTMNODAM	0.6	y	Water residence time (no dam)	Field measurement and Opper 1982
HYDRESTMDAM	0.8	y	Water residence time (with dam)	Ibid.
LAKEBED	18.57	m (msl)	Mean initial elevation of sediment-water interface	Field estimate
BULKDENS	30000	$g\ m^{-3}$	Bulk density of Surfsed	Field measurement
BEDDENS	500000	$g\ m^{-3}$	Bulk density of Permsed	Field measurement
SDR	0.5	$\%\ y^{-1}$	Specific rate of deposition	Calibrated from field measurement
SVELP	300	$m\ y^{-1}$	Settling velocity of plankton	Estimate
SVELD	2523	$m\ y^{-1}$	Settling velocity of detritus	Sheng and Lick 1979; Lick 1982
SVELI	2523	$m\ y^{-1}$	Settling velocity of inorganic particles	Ibid.
α	36.85	$\%\ y^{-1}$	Model coefficient (for SRR)	Calculated using Somlyody 1982
γ	-0.084	$\%\ m^{-1}$	Model coefficient (for SRR)	Ibid.

Table 5-1--continued

Constant	Value	Units	Description	Source of value
MAXPAR	16	cal $cm^{-2}h^{-1}$	Maximum photosynthetically active radiation at mean depth	Estimated using Gates 1963
OPTPAR	8	cal $cm^{-2}h^{-1}$	Optimum photosynthetically active radiation at mean depth	Estimate
β	0.069	% $g^{-1}m^3$	Extinction coefficient	Curve-fitted using Beeton, 1958
MAXBENTHICBACT	15.5	$g_{Srp}\ g_{Pbenth\text{-}bact}^{-1}y^{-1}$	Max. P-uptake rate for benthic bacteria	Estimate
MAXPLANKBACT	15.5	$g_{Srp}\ g_{Pplank\text{-}bact}^{-1}y^{-1}$	Max. P-uptake rate for planktonic bacteria	Estimate
MAXBENTHICAUTO	3	$g_{Srp}\ g_{Pbenth\text{-}alg}^{-1}y^{-1}$	Max. P-uptake rate for benthic algae	Estimate
MAXPLANKAUTO	10	$g_{Srp}\ g_{Pplank\text{-}alg}^{-1}y^{-1}$	Max. P-uptake rate for planktonic algae	Estimate
KM_1	0.31	$g_{Srp}\ m^{-3}$	Half saturation constant for P-uptake rate by benthic bacteria	Fuhs et al. 1972
KM_2	0.036	$g_{Srp}\ m^{-3}$	Half saturation constant for P-uptake rate by benthic algae	Fuhs et al. 1972
KM_3	0.31	$g_{Srp}\ m^{-3}$	Half saturation constant for P-uptake rate by planktonic bacteria	Fuhs et al. 1972
KM_4	0.036	$g_{Srp}\ m^{-3}$	Half saturation constant for P-uptake rate by planktonic algae	Fuhs et al. 1972
BACTFRAC	0.50	$g_{Bact}\ g_{Plankton}^{-1}$	Bacterial fraction of plankton	Saunders et al. 1980
AUTOFRAC	0.42	$g_{Algae}\ g_{Plankton}^{-1}$	Algal fraction of plankton	Calculated from field data
EUKFRAC	0.08	$g_{Eukhet}\ g_{Plankton}^{-1}$	Eukaryotic heterotroph fraction of plankton	Bays and Crisman, 1983
PBACTCONT	0.008	$g_P\ g_{Bacteria}^{-1}$	P-content of bacterio-plankton	Redfield et al. 1963; Wetzel, 1983
PAUTOCONT	0.004	$g_P\ g_{Algae}^{-1}$	P-content of planktonic algae	Ibid.
PEUKCONT	0.004	$g_P\ g_{Eukhet}^{-1}$	P-content of eukaryotic-heterotroph plankton	Ibid.

Table 5-1--continued

Constant	Value	Units	Description	Source of value
PDETRITUSCONT	0.002	$g_P\ g_{Detritus}^{-1}$	P-content of suspended detritus	Redfield et al. 1963; Wetzel 1983
PINORGCONT	0.004	$g_P\ g_{Inorgseston}^{-1}$	P-content of inorganic seston	Ibid.
BENTHICBACTFRAC	0.75	$g_{Bact}\ g_{Benthos}^{-1}$	Bacterial fraction of benthos	Wetzel, 1983
BENTHICAUTOFRAC	0.20	$g_{Algae}\ g_{Benthos}^{-1}$	Algal fraction of benthos	Estimate
BENTHICEUKFRAC	0.05	$g_{Eukhet}\ g_{Benthos}^{-1}$	Eukaryotic-heterotroph fraction of benthos	Estimate
PBENTHICBACTCONT	0.012	$g_P\ g_{Bacteria}^{-1}$	P-content of bacterial-benthos	Redfield et al. 1963; Wetzel 1983
PBENTHICAUTOCONT	0.004	$g_P\ g_{Algae}^{-1}$	P-content of benthic-algae	Ibid.
PBENTHICEUKCONT	0.004	$g_P\ g_{Eukhet}^{-1}$	P-content of eukaryotic-heterotroph benthos	Ibid.
PDETSEDCONT	0.002	$g_P\ g_{Detsed}^{-1}$	P-content of detrital surfsed	Ibid.
PINORGSEDCONT	0.003	$g_P\ g_{Inorgsed}^{-1}$	P-content of inorganic surfsed	Ibid.
K_f	0.045		Coefficient (for FILTER)	Estimate
PLOAD	0.05	$g_{Srp}\ m^{-2}y^{-1}$	Srp contribution from runoff	Gottgens and Montague, 1987; Baker et al., 1981
Q	1350	$g_{Litpart}\ m^{-2}y^{-1}$	Gain in LITPART from production and influx	Estimate
K_t	0.02		Coefficient (for TRAP)	Estimate
LITRESTM	0.15	y	Litpart residence time	Estimate
M	0.003		Slope coefficient for specific export rate	Estimate
B_2	0.007	$\%\ y^{-1}$	Y-intercept for specific export rate	Estimate

Table 5-1--continued

Constant	Value	Units	Description	Source of value
SUBMLITAREA	6 E6	m^2	Submerged littoral area	Calculated from lake bathymetry
LAKEVOLUME	40.5 E6	m^3	Lake volume	Ibid.
SLSR	0.005	% y^{-1}	Specific littoral sedimentation rate	Estimate
HALFLIFE	5	y	Veg. littoral area residence time	Calibrated to proposed littoral area dynamics
B_3	64.42		Model coefficient for gain in veglitarea	Ibid.
N	-5.07		Model coefficient for gain in veglitarea	Ibid.
PLITDETRITUSCONT	0.002	$g_P\ g_{Litdetritus}^{-1}$	P-content of detrital Litpart	Redfield et al. 1963; Wetzel 1983
PLITINORGCONT	0.004	$g_P\ g_{Litinorg}^{-1}$	P-content of inorganic Litpart	Ibid.
PLITPLANKTONCONT	0.006	$g_P\ g_{Litplankton}^{-1}$	P-content of planktonic Litpart	Ibid.
DETRITUSLITFRAC	0.2	$g_{Detritus}\ g_{Litpart}^{-1}$	Detrital fraction of Litpart	Estimate
INORGLITFRAC	0.2	$g_{Inorg}\ g_{Litpart}^{-1}$	Inorganic fraction of Litpart	Estimate
PLANKTONLITFRAC	0.6	$g_{Plankton}\ g_{Litpart}^{-1}$	Planktonic fraction of Litpart	Estimate
PDELLITDETRITUSCONT	0.002	$g_P\ g_{Dellitdetritus}^{-1}$	P-content of detrital Litpart	Redfield et al. 1963; Wetzel 1983
PDELLITINORGCONT	0.004	$g_P\ g_{Dellitinorg}^{-1}$	P-content of inorganic Litpart	Ibid.
PDELLITPLANKTONCONT	0.006	$g_P\ g_{Dellitplankton}^{-1}$	P-content of planktonic Litpart	Ibid.
DETRITUSDELLITFRAC	0.2	$g_{Detr}\ g_{Dellitpart}^{-1}$	Detrital fraction of Litpart	Estimate
INORGDELLITFRAC	0.2	$g_{Inorg}\ g_{Dellitpart}^{-1}$	Inorganic fraction of Litpart	Estimate

Table 5-1--continued

Constant	Value	Units	Description	Source of value
PLANKTONDELLITFRAC	0.6	$g_{Plank}\ g_{Dellitpart}^{-1}$	Planktonic fraction of Litpart	Estimate
V_1	500	$g_{Srpwater}\ g_{Pinorg}^{-1}$	Specific reaction rate for Pinorg → Srpwater	Estimate
V_2	500	$g_{Srppore}\ g_{Pinorgsed}^{-1}$	Specific reaction rate for Pinorgsed → Srppore	Estimate
KR_1	2.67	$g_{Pinorg}\ g_{Srpwater}^{-1}$	Equilibrium constant for Pinorg → Srpwater	Calculated from initial values
KR_2	3.06	$g_{Pinorgsed}\ g_{Srppore}^{-1}$	Equilibrium constant for Pinorgsed → Srppore	Ibid.
PLANKSPECREGRT	500	$g_{Srp}\ g_{Eukhet}^{-1}y^{-1}$	Specific regeneration rate of Srp by euk.het. plankton	Estimated using Ferrante, 1976
BENTHOSSPECREGRT	500	$g_{Srp}\ g_{Eukhet}^{-1}y^{-1}$	Specific regeneration rate of Srp by euk.het. benthos	Ibid.
PLIFETM	0.05	y	Average lifetime of plankton	cf. Wetzel, 1983
BLIFETM	0.025	y	Average lifetime of benthos	Ibid.
KDETRITUS	20	$g_{Pplank}\ g_{Pdetr}^{-1}y^{-1}$	Specific rate of plankton decomposition	Estimate
KDETSED	20	$g_{Pbenth}\ g_{Pdetsed}^{-1}y^{-1}$	Specific rate of benthos decomposition	Ibid.

per m^3 per year, respectively. Areal rates are converted to volumetric rates by dividing by water-column depth.

The dam influences the modeled relationships in three ways (Figure 5-3).

(1) It reduces the water flow rate through the lake by increasing the water residence time (τ_w).

$$\tau_w = \frac{\text{Lake Volume}}{\sum \text{Outflows}} = \frac{\text{Lake Volume}}{E_t + O_{sw} + O_{gw}} \qquad (5\text{-}1)$$

E_t (evapo-transpiration) averaged 106.849 m^3/day for Newnan's Lake, using coefficients from Opper (1982) from nearby Lake Wauburg. O_{sw} (surface water outflow) was computed from stage-discharge relationships developed during 1989 when the spillway was removed for a period of 3 months. The spillway reduced mean O_{sw} from 103.680 to 51.840 m^3/day (Gottgens and Crisman 1991). O_{gw} (groundwater outflow) was calculated using Walton's (1970) equation for downward leakage (Q_c);

$$Q_c = \frac{\rho}{m} \times \Delta h \qquad (5\text{-}2)$$

where ρ is the coefficient of permeability of deposits, m is the saturated thickness of the lake bottom sediments and Δh represents the average difference between the lake level and the potentiometric surface of the Floridan aquifer. Following work by Opper (1982), estimates of 1280 m^3/day, 3 m, and 4 m were used, respectively, for ρ, m, and Δh. This produces a Q_c estimate of 1706 m^3/day. Substituting values for E_t, O_{sw}, and Q_c into equation (5-1) yields a τ_w of 0.6 years for the lake without spillway and 0.8 years with the spillway in place. This increase in water residence time enhances settling of suspended matter. It also increases lake volume, which results in a larger surface area and hydraulic head pressure. Therefore, rates of evapo-transpiration and losses to groundwater increase. Both involve loss of water without loss of particulate material. The result is increased accumulation of flocculent bottom material.

(2) The spillway reduces flushing of suspended matter by eliminating discharge of bottom-water. This layer of water is rich in solids due to sediment resuspended by wind and wave-induced currents (Sheng and Lick 1979). Outflow over the dam is restricted to less turbid surface waters. The concentration of suspended particulate matter in the surface outflow of Newnan's Lake averaged 12.5 mg/l (n=22) during 45 days immediately prior to removal of the spillway in 1989 and 18.0 mg/l (n=8) for the same length of time after the stoplogs were removed (Gottgens and Crisman 1991). Flushing of dissolved material (such as PO_4) is not affected by bottom-water drainage and is simply a function of water residence time and P concentration.

(3) The spillway increases the lake stage and reduces the amplitude of the seasonal water-level fluctuations. The Newnan's Lake stage hydrograph from 1945 to present has stages recorded between 19.51 and 21.77 m (msl) (Adkins 1991). This record is complete with the exception of the period 1954-57 and 1962-64. During these times severe droughts reduced the lake levels in all the lakes in the Orange Creek basin. Average stage for the prespillway period (1945-1967) was 20.24 m (msl) with an average annual amplitude of oscillation of 0.55 m (Skoglund 1990; Adkins 1991). For the postspillway period (1967-1990) these numbers were respectively 20.37 and 0.39. Higher lake stage and smaller amplitude of water-level fluctuation both increase water depth which, in turn, reduces the shear stress from wind-wave action on surface sediments. Consequently, the magnitude of resuspension of this material decreases.

Surface sediments are operationally defined in the model as the layer of material immediately below the sediment-water interface involved in the settling and resuspension cycle. Based on examination of bottom cores from Newnan's Lake, the thickness of this "fluid" zone was estimated between 3 and 7 cm. The actual thickness of this layer may depend on the intensity and frequency of wave action, water depth, and the bulk density of the bottom material. The initial value of 4.5 cm, used in this study, corresponds with values reported by Hwang and Mehta (1989) for similar sediment in a non-stratified Florida Lake. An increase in the amount of surficial, flocculent sediment will increase resuspension. More suspended particles (seston) lead to increased settling rates and, in turn, more surface sediments. This positive feedback loop is illustrated for the inorganic, live organic (benthos and plankton), and dead organic (detritus) fractions in Figure 5-3 (loops 1, 2,and 3). The effect of each of these positive loops may be bounded by negative loops between seston and settling (loops 4, 5, and 6) and between surface sediments and resuspension (loops 7, 8, and 9). Core profile data were used to partition surface sediments in an organic and inorganic fraction (0.66 and 0.34, respectively) to include the different P dynamics (e.g. uptake, sorption, decomposition, regeneration) and settling velocities of each fraction in the hypothesis.

The P content of each fraction was initially estimated using Redfield et al. (1963), whereby 1 gram of plankton contains 0.012 grams of P, assuming the carbon content of the plankton is 50%. When back-calculated from a water-column P concentration of 0.08 g/m^3 (averages from Shannon 1970 and Canfield 1981) and an average seston concentration of 20 g/m^3 (field data), a much lower P-content of 0.004 g_P/g was found. If 80% of the total seston is organic (Chapter 3) and half of this fraction is assumed to be planktonic (i.e. 8 grams of plankton per m^3 and 8 grams of detrital seston per m^3), a P content of 0.006 g_P/g for plankton, 0.002 g_P/g for detritus, and 0.004 g_P/g for the remaining 4 grams of inorganic seston per m^3 yield a water-column P concentration of 0.08 g/m^3. These P-content values correspond with field data discussed by Fuhs et al. (1972) and Wetzel (1983) in which C:P atom-ratios in excess of 200:1, i.e. approximately double the ratio proposed by Redfield, are encountered commonly.

Settling velocities for the inorganic and detrital fractions were taken from Sheng and Lick (1979). These velocities may range over more than four orders of magnitude depending on particle size, concentration and mineral composition of the suspended sediments, and ionic strength of the water (Fukuda and Lick 1980; Lee et al. 1981). In this model, settling rate was assumed to be proportional to the concentration of seston (Lick 1982; Hwang and Mehta 1989). Plankton settling was estimated one order of magnitude smaller, due to characteristics that improve flotation or reduce sinking rates. Examples of these characteristics are density reduction by the production of mucilage sheaths (most blue-green phytoplankton) or gas vacuoles (bacteria and algae), as well as active swimming (eukaryotic heterotrophs).

Resuspension of bottom material due to wind-driven currents was assumed to be negligible in the profundal zone of the lake (Lam and Jaquet 1976; Luettich et al. 1990). All entrainment occurs as a result of oscillations produced in the water column by wind-wave action at the water surface. These surface waves produce a series of oscillating motions downward in the water column which may extend to the sediment-water interface in shallow lakes (Pollman 1983). The height of vertical oscillations attenuates rapidly with depth with an approximate halving of the cycloid diameter for every depth increase of $\lambda/9$, where λ is the wavelength of the surface waves (Wetzel 1983). Near the sediment boundary layer, the oscillatory motion is reduced to a simple reciprocating horizontal motion (U.S. Army Coastal Engineering Research Center 1977). When the shear force of this displacement exceeds the bulk shear strength of the surficial sediment deposits, sediment resuspension occurs (Terwindt 1977; Hwang and Mehta 1989). When λ and water depth are known, this displacement can be calculated. While not directly proportional to wave height (H), a ratio of 1:20 for $H:\lambda$ is a common average (Wetzel 1983). H is generally proportional to the square root of the fetch (x), or distance over which the wind blows uninterrupted by land. The height of the highest waves observed is approximated by

$$H = 0.105\sqrt{x} \qquad \text{(Wetzel 1983)} \quad (5\text{-}3)$$

Average fetch for Newnan's Lake is 3.5 km. Hence, maximum wave height is 0.62 m and λ is 12.4 m. Halving the cycloid diameter for every depth increase of $\lambda/9$ yields a vertical displacement of 0.28 m at the sediment-water interface in Newnan's Lake, with an average water depth of 1.5 m. Field data from a similar shallow, exposed lake (Somlyódy 1982) with a mean displacement of 0.24 m at the boundary layer demonstrated a resuspension flux of $12{,}410 \times W$ g $m^{-2}y^{-1}$, where W is the wind speed (m/sec). When applied to Newnan's Lake a flux of 44,200 g $m^{-2}y^{-1}$ is computed, assuming an average wind speed of 3 m/sec. Resuspension is assumed to be proportional to the amount of surficial sediment, although it may be influenced by composition (particle size, organic matter content), water content, and activity of benthos (Lee et al. 1981). Surface sediment was estimated at 1,300 g/m^2 (using field data on bulk density and the thickness of this layer). Therefore, if the total resuspension flux equals 44,200 g $m^{-2}y^{-1}$, the specific rate of resuspension (SRR) equals 34 g $g^{-1}y^{-1}$. This value for SRR applies to conditions of maximum wave height and a water depth of 1.5 m. Since the model simulates resuspension under average wave height, I applied this value of SRR to estimate resuspension under conditions of the lowest water level (i.e highest shear stress). In the model, SRR varies with water depth (m) according to

$$SRR = \alpha \times e^{\gamma \times Depth} \tag{5-4}$$

Values of 36.85 and -0.084 for the model coefficients α and γ, respectively, result in an SRR value of 34 g $g^{-1}y^{-1}$ at minimum water depth and plausible resuspension rates with increasing water depth. The relationship between SRR and water depth is graphed in Figure 5-4a. Entrainment of pore-fluid soluble-reactive phosphorus (SRP) is assumed to occur at the same specific rate (Figure 5-3, loop 10). Water-column SRP only settles after incorporation into organic or inorganic seston.

Surface sediment is lost downward to permanent sedimentation. For each fraction, this loss rate is proportional to the amount of surface material present (Figure 5-3, loops 11, 12, and 13). The model equations for sedimentation were calibrated to correspond with an average prespillway sedimentation rate of 650 g/m^2 calculated from ^{210}Pb dated profundal cores. No downward loss of pore-fluid SRP is assumed. Instead, this phosphorus may be first absorbed to inorganic particles or taken up by benthos after which sedimentation as particulate matter may occur.

Direct absorption (and desorption) of SRP from sedimentary detritus is assumed to be negligible. The bacterial coating of detrital particles, a component of the

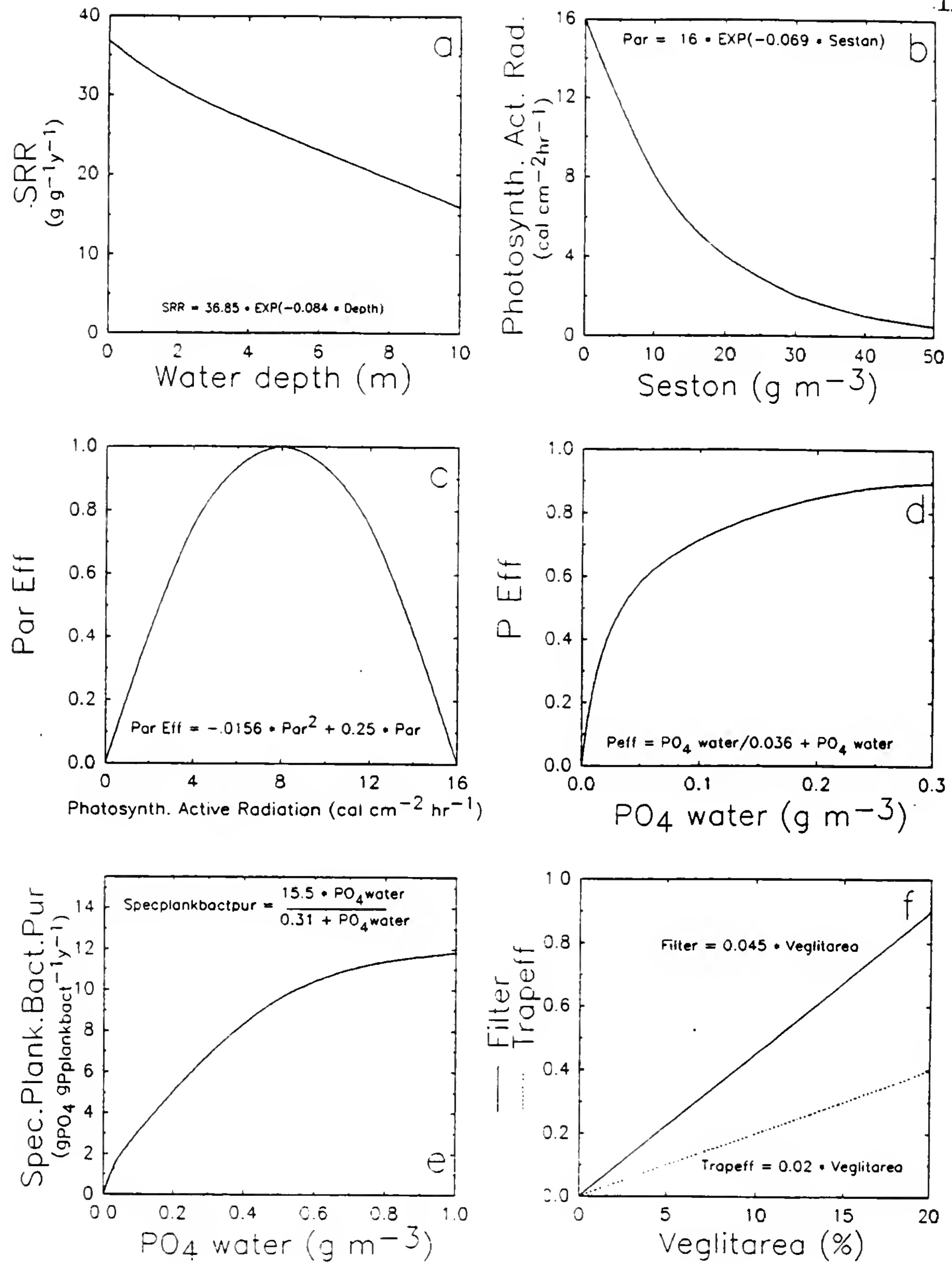


Figure 5-4. Plots of seven hypothesized relationships used in the model.

benthos, likely traps the large majority of SRP. P derived by the bacteria from detritus may flow back to pore-fluid SRP via P regeneration. The same pathways are assumed to exist in the water column.

Absorption and desorption processes of P between inorganic sediment and interstitial pore-fluid (Figure 5-3, loops 14 and 15) and between inorganic seston and water (loops 16 and 17) are modeled as a function of a specific reaction rate and the extent to which the respective P concentrations deviate from equilibrium conditions. For example, the rate of P desorption from inorganic seston to the surrounding water (in $g_{Srpwater}\ m^{-2}y^{-1}$) can be expressed as

$$Sorp = v \times (P_{inorg} - K_R \times Srp_{water}) \tag{5-5}$$

where v is the specific reaction rate ($g_{Srpwater}\ g_{Pinorg}{}^{-1}y^{-1}$) and K_R is the equilibrium constant. At equilibrium,

$$K_R = \frac{P_{inorg}}{Srp_{water}} \tag{5-6}$$

Using initial model values for PINORG and SRPWATER, the equilibrium constant for this reaction equals 2.67. Analogously, the K_R value for the sorption reaction between P in inorganic sediment and pore-fluid equals 3.06.

Feedback loops involved in interactions between plankton, water-column SRP, and detrital seston are made more apparent in Figure 5-5. The same influences are hypothesized in the model for interactions between benthos, pore-fluid SRP, and sedimentary detritus. The difference is the absence of light. This reduces the rate of P uptake in benthic algae, a process which requires energy normally supplied by photosynthesis (Kuenzler and Ketchum 1962). To prevent redundancy, the following description is limited to the feedback relationships between plankton, SRP, and detrital seston in the water column.

In this model, plankton consist of bacteria, algae, and eukaryotic heterotrophs. Since the model emphasizes P dynamics, these fractions were treated separately. Each of the fractions may affect the flow of P through the system in a different way. P is taken up by both bacteria and algae, while being regenerated by eukaryotic heterotrophs. P is recycled from the detritus by bacteria and eukaryotic heterotrophs. Bacteria, however, fix rather than mineralize P (Day et al. 1989). Therefore, no direct P regeneration to the water column from bacteria is assumed to occur. Direct P regeneration to the water does occur through ingestion and subsequent excretion by eukaryotic heterotrophs. Literature values for similar lakes and field data from this study were used to obtain the relative size of each fraction. Again, a plankton concentration of

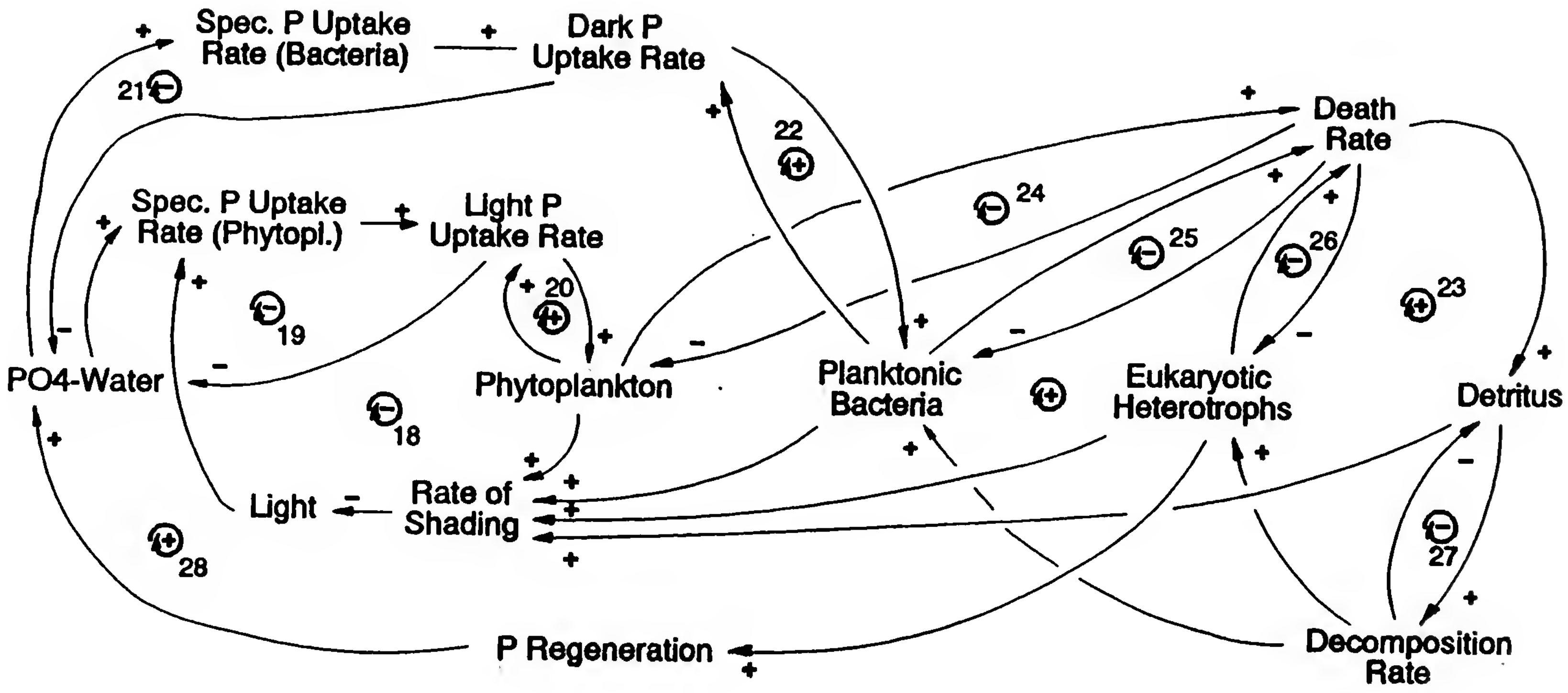


Figure 5-5. Plankton-phosphorus sector of the system influence diagram for the profundal zone showing additional detail in the model hypothesis.

8 g/m^3 was used. Bacterial biomass was estimated 4 g/m^3 (Saunders et al. 1980). An algal biomass of 3.35 g/m^3 was derived using field data on average chlorophyll a in Newnan's Lake and assuming that this pigment constitutes 1.5% of algal biomass (A.P.H.A. 1985). The remaining 0.65 g/m^3 is then assumed to be eukaryotic heterotrophs. This corresponds with data reported by Bays and Crisman (1983) for the productive Florida lakes from their zooplankton-biomass study. Bacterial, algal, and eukaryotic heterotroph fractions were, therefore, respectively 0.5, 0.42, and 0.08.

Specific P-uptake by phytoplankton is affected by light (e.g. photosynthetically active radiation, PAR) and concentration of SRP in the water column (Fuhs et al. 1972). More suspended matter in the water column reduces PAR available for photosynthesis which lowers the specific P-uptake rate by phytoplankton (Figure 5-5, loop 18). Light reduction by seston is assumed to be independent of the type of seston (i.e. inorganic, detrital or planktonic). A reduced P-uptake rate leaves more dissolved P (loop 19). It also yields less phytoplankton which, in turn, depresses P-uptake (loop 20). As seston concentration (g/m^3) increases, available PAR (cal cm^{-2}hr^{-1}, at mean water depth) declines according to

$$PAR = 16 \times e^{-0.069 \times Seston} \tag{5-7}$$

where 16 is the value of PAR_{max} (Gates 1963) and -0.069 is the extinction coefficient (% per g/m^3 of seston), estimated using data from Beeton (1958). This relationship is shown in Figure 5-4b. A hypothesized effect of PAR on the specific P-uptake rate of phytoplankton (PAREFF) is illustrated in Figure 5-4c. Light of high intensity is detrimental to many algae and is particularly associated with photo-oxidative destruction of enzymes involved in photosynthesis (Steemann Nielsen 1962). An intensity of 8 cal $cm^{-2}hr^{-1}$ was used in the model as optimum PAR.

The hypothesized effect of SRP concentration on specific P-uptake by phytoplankton (PEFF) is shown in Figure 5-4d. It follows Michaelis-Menten kinetics with a half-saturation constant of 0.036 g_{SRP}/m^3 (Fuhs et al. 1972).

The specific P-uptake rate, v′ ($g_{SRP}\ g_{Pplankton}^{-1}y^{-1}$), may then be given by

$$v' = v'_{max} \times \sqrt{(PAREFF \times PEFF)} \tag{5-8}$$

Thus, the specific P-uptake rate is zero in the absence of light or nutrients and the maximum rate only occurs when both the light and nutrient regime are optimal. Fuhs et al. (1972) found v'_{max} in excess of 1000 $g_{SRP}\ g_{Pplankton}^{-1}y^{-1}$ in a detailed study of P kinetics in P-starved planktonic diatoms. They cautioned, however, that these laboratory results from a few diatom species should not be generalized to algal assemblages in the field. Turnover rates (i.e. specific uptake rates) of phosphorus have been studied in lakes of different productivity and at different times of the year. The reported rates vary by several orders of magnitude and are faster under oligotrophic conditions. Peters (1975) reported P turnover rates ranging from 7 to 131,000 g $g^{-1}y^{-1}$ for epilimnia of 15 European lakes. With no literature values specific for the conditions in Newnan's Lake, v'_{max} was estimated at 10 $g_{SRP}\ g_{Pplankton}^{-1}y^{-1}$. This guess was based on the low turnover rates found by Peters (1975) for systems with high primary productivity, similar to Newnan's Lake. The specific P-uptake rate multiplied by the algal-P concentration (g_P/m^3) yields the P-uptake rate ($g_{SRP}\ m^{-3}y^{-1}$).

Estimates for the v'_{max} for planktonic bacteria, benthic bacteria, and benthic algae were based on the value used for v'_{max} for phytoplankton. Because bacteria are capable of storing considerable P (represented in the model by an elevated P-content) their rates will be higher relative to phytoplankton specific P-uptake (Fuhs et al. 1972). The specific P-uptake rate for benthic algae is low due to the absence of light (Kuhl 1974). Michaelis-Menten kinetics were used to describe the increase in specific P-uptake rate by planktonic bacteria with increasing SRP in the water column (Figure 5-4e). The resulting increase in P-uptake rate, in turn, reduces available SRP (Figure 5-5, loop 21). It also yields more planktonic bacteria, which stimulates additional P-uptake (loop 22).

Components of the plankton and detrital seston are linked in a positive feedback loop. As plankton increases, death rates increase proportionally and produce more detritus. This leads to higher rates of decomposition and an increase in decomposer biomass (loop 23). Decomposers include bacteria and eukaryotic heterotrophs. The effect of this positive loop is moderated by negative loops between components of the phytoplankton and death rate (loops 24, 25, and 26) and between detritus and decomposition rate (loop 27). Average lifetime (for all fractions of plankton combined) was estimated to be 0.05 years (using Wetzel 1983). This lifetime was halved for the benthos because of the larger fraction of short-lived microplankton (i.e. bacteria). A specific rate of decomposition of 20 (g $g^{-1}y^{-1}$) was used, indicating a turnover time of approximately 18 days. Such high rates are plausible under conditions of year-round warm temperatures and correspond with data reported in Wetzel (1983).

Mineralization of P occurs during egestion and excretion by eukaryotic heterotroph consumers (Figure 5-5, loop 28). A regeneration rate of 50 $g_{SRP}\ g_{Plankton}{}^{-1}y^{-1}$ was reported by Ferrante (1976) for a mixed epilimnetic zooplankton assemblage in Lake Michigan with water temperatures between 3 and 24°C. As an approximation, a value of 500 $g_{SRP}\ g_{Plankton}{}^{-1}y^{-1}$ was used to account for higher year-round temperatures and smaller-sized zooplankton in Newnan's Lake (Crisman 1986a).

Littoral. Causal feedback relationships hypothesized for the littoral zone of the lake are shown in Figure 5-6. The outflows from this zone connect with the corresponding components in the profundal zone. The emphasis in the littoral influence diagram is on the effect of the dam on the flow of particles and phosphorus from the littoral to the profundal area of the lake. As discussed earlier, the reduction in amplitude of annual water-level fluctuations, resulting from the dam, may gradually reduce the extent of the vegetated littoral zone along the periphery of the lake. This may reduce the ability of this zone of plants and associated epiphytes to filter and trap phosphorus and sediments carried into the lake from the surrounding watershed (Mickle and Wetzel 1978). This, in turn, may contribute to enhanced primary production in the lake.

As the amplitude of annual water-level fluctuations decreases from 0.55 to 0.39 m with the installation of the

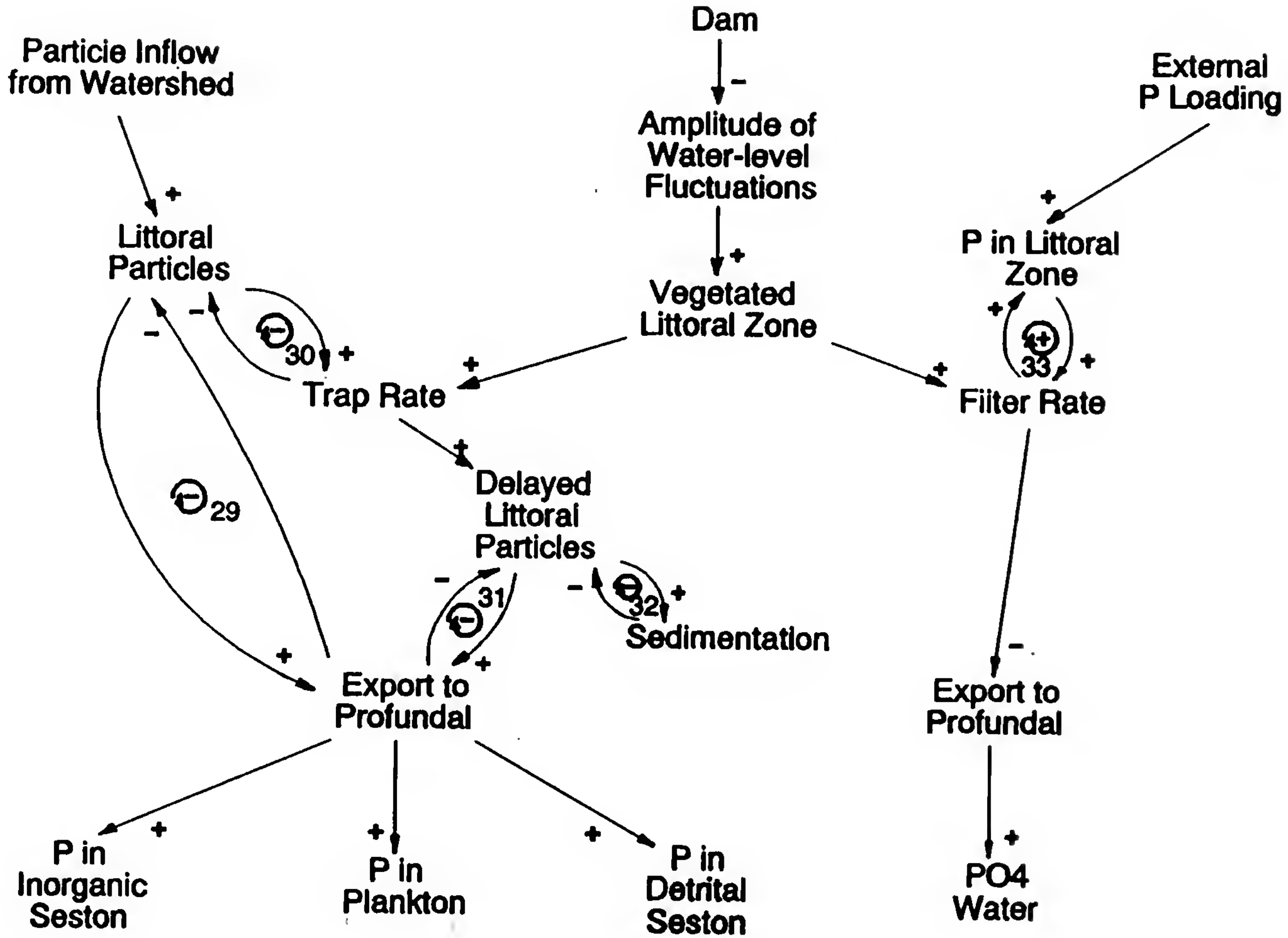


Figure 5-6. Influence diagram of the modeled hypothesis for the littoral zone. See text for feedback-loop descriptions.

dam, the wetland fringe in the lake should be reduced from 20% to perhaps 10% of lake surface area. This reduction is modeled with a halflife of 5 years, i.e. 5 years after the dam is installed 15% of the lake surface area remains as vegetated littoral zone. The extent of this vegetated zone (y, in % of lake surface area) as a function of amplitude of water-level fluctuation (x, in m) is modeled as a sigmoid relationship

$$y = \frac{1}{1 + 64.42 \times e^{-5.07\,x}} \tag{5-9}$$

This equation was selected because it generates plausible values for the wetland fringe of 2% with no water-level fluctuations and 50% when the amplitude of fluctuation approximates half of the mean depth of the lake. In addition, it generates the proposed extent of wetland fringes under pre- and postspillway conditions discussed above.

Littoral particles brought into the lake from the surrounding watershed may either be temporarily trapped in littoral vegetation (Figure 5-6, loop 30) or may pass through this zone (loop 29). Trapped particles erode from littoral substrate after a delay time (0.15 years) and move to deeper areas of the lake (loop 31), while a small fraction is lost to sedimentation (loop 32). Both the rate of trapping and the rate at which P is filtered from the overlying water by submersed littoral vegetation and epiphytes (loop 33) are assumed to be proportional to the extent of vegetated littoral area (Figure 5-4f).

In summary, the dam in the outlet of Newnan's Lake influences a system of feedback relationships in the lake through increased water residence time, and reductions in bottom-water drainage and amplitude of annual water-level fluctuations. These influences contribute to the lake management problems described in the introduction of this chapter. The relationships in this hypothesis are based on data from field and laboratory work or are independently derived from the literature. In the absence of data or theory, relationships are based on estimate or opinion. Many assumptions have been stated about unknown parameter values and equation forms. The following simulation and model analysis subjects the assumptions to tests that attempt to falsify the feedback hypothesis, identify parameters to which the model is sensitive, and compare the effect of changes in the model with similar changes made in the field.

Results and Discussion

Standard Run Output

The output from the simulation is shown for five model variables (Figure 5-7). These variables were chosen because of their potential significance for lake management

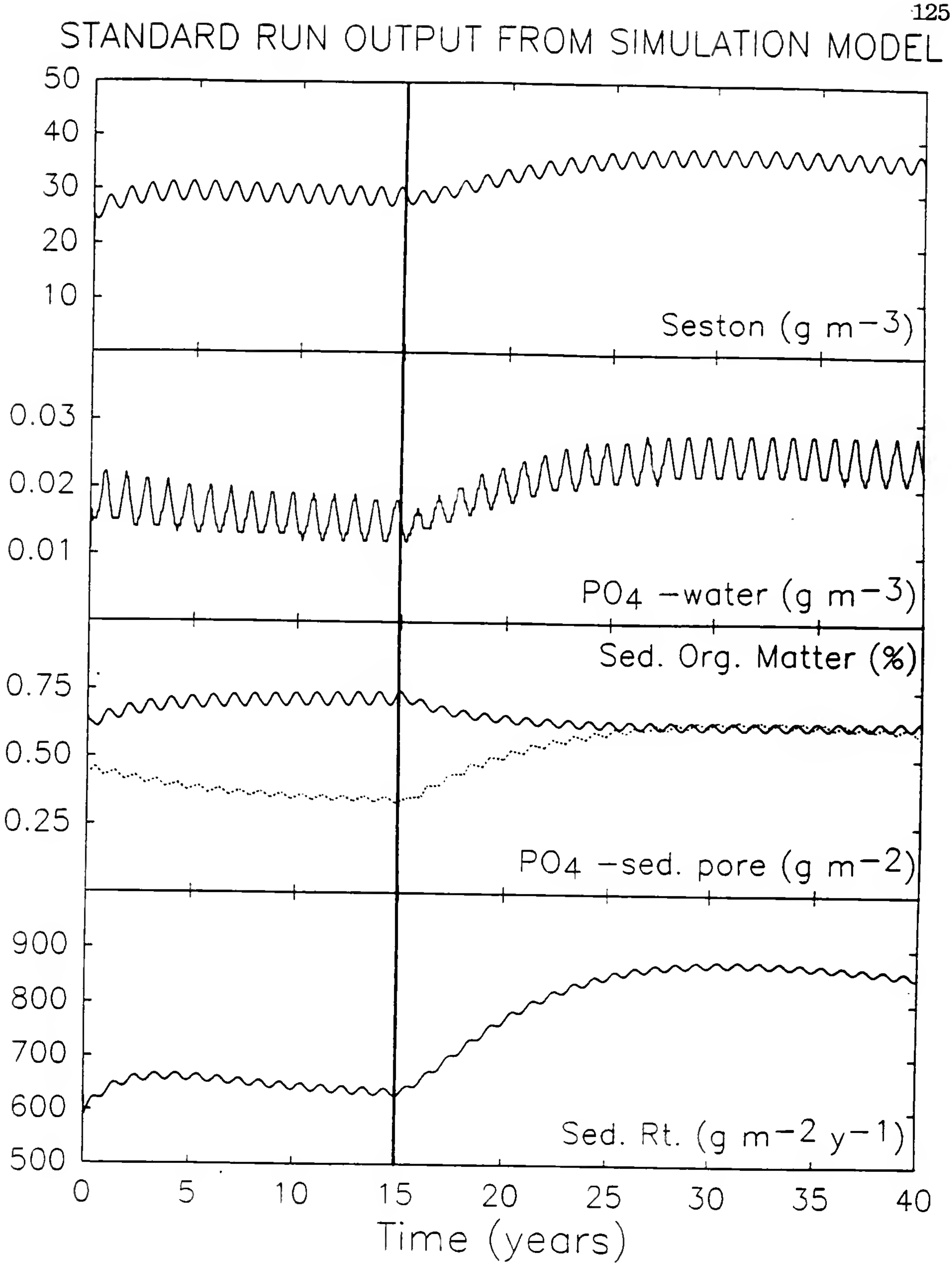


Figure 5-7. Standard run output from simulation model with spillway installation at 15 years.

(e.g seston, water-column SRP) and because they may be compared to collected field data (e.g. sedimentary P and organic matter content, sedimentation rate). The simulation represented 40 years with spillway installation initiated after 15 years. Dependent variables were scaled individually as indicated on the vertical axis. An integration interval of 0.0005 years (of simulated time) was selected to avoid model instability and serious integration error, while minimizing computer run-time. The selected interval was 1/4 of the inverse of the highest specific rate used in the model. The sinusoid pattern of the output was produced by the simulated annual water-level fluctuations.

Prespillway seston concentrations averaged 28.8 g/m^3 and increased by 19% after spillway installation. The model output reflected the hypothesis that the dam increases concentration of suspended solids. Separate analysis of the pre- and postspillway dynamics of the inorganic, planktonic, and detrital fractions did not show implausible shifts in the composition of the seston (Table 5-2). The ratios of the average concentrations of inorganic, planktonic, and detrital seston were 1 : 2.4 : 2.1 and 1 : 2.0 : 1.8 pre- and postspillway, respectively.

Soluble P concentrations in the water column increased from an average of 0.016 g/m^3 prespillway to 0.028 g/m^3 approximately 10 years postspillway installation after which the average remained constant (Figure 5-7). This corresponded to the pattern for the other model output variables shown for the standard run, with the exception of sedimentary organic matter content. The concentration range for water-column SRP compared well with reported ranges of 0.003-0.028 (Shannon 1970) and 0.01-0.06 g/m^3 (Crisman 1986a).

Table 5-2. Pre- and postspillway concentrations of inorganic, planktonic, detrital, and total seston from standard model run. Values given are the average, minimum, and maximum (in g/m^3); time-of-minimum and time-of-maximum (in years after model start).

	Prespillway					Postspillway				
	Avg	Min	Tmin	Max	Tmax	Avg	Min	Tmin	Max	Tmax
Inorganic seston	5.3	4.0	0.0	6.5	0.6	7.3	4.4	15.0	8.2	30.9
Planktonic seston	12.6	8.0	0.0	14.4	6.8	14.6	11.6	16.3	17.1	32.1
Detrital seston	10.9	8.0	0.0	11.7	5.9	12.9	10.7	16.3	14.0	32.1
Total seston	28.8	20.0	0.0	31.3	4.8	34.8	27.4	15.3	38.3	31.8

Organic matter content of the surface sediment, computed in the model as $(g_{Benthos}+g_{Detritus})/g_{Surf.sed.}$, decreased after spillway installation. This was contrary to in situ measurements, which showed higher organic matter content in recent sediments (Chapter 2), assuming no decomposition of organic matter after permanent sedimentation (Gale et al. 1992). Regardless of the validity of this assumption, a reduction in the organic matter content of the surface sediments is unlikely when increased water-column SRP concentration probably elevates primary production. The reduction was produced in the model by a 44% increase in the inorganic fraction of surface sediment compared with 15% increases in benthos and detritus following spillway installation. This inorganic fraction was computed in the model by dividing the ratio of P in inorganic sediment (g_P/m^2) and the P content of inorganic sediment ($g_P/g_{Inorg.sed.}$) by the total amount of surface sediments (g/m^2). Decreasing the value of the equilibrium constant (K_R2) in the sorption reaction between P in inorganic sediment and P in sedimentary pore-fluid from 3.06 to 2.67 (i.e. equal to the equilibrium constant for the reaction between P in inorganic seston and water-column SRP) eliminated the decrease in sedimentary organic matter postspillway without significantly affecting other model output. K_R2 represented the ratio at equilibrium of P in inorganic sediments and SRP in interstitial pore-fluid. In isotopic exchange studies, using ^{32}P as a tracer, Li et al. (1972) found values for this equilibrium constant ranging from 1.33 to 4.26 for four Wisconsin lakes. Thus, decreasing K_R2 from 3.06 to 2.67 kept this equilibrium constant within a plausible value range.

Soluble P concentration in the interstitial pore-fluid doubled after spillway installation (Figure 5-7). No implausible shifts in the composition of total sedimentary phosphorus occurred (Table 5-3). The ratio of average P concentration in inorganic sediment, benthos, detrital sediment, and pore-fluid was 3.2 : 2.2 : 4.5 : 1 prespillway and 3.1 : 1.7 : 3.5 : 1 after spillway installation.

In order to compare simulated total sedimentary P concentrations with those measured in the cores, the former needed to be converted to $g_P/g_{Surf.sed.}$. Using the average simulation values for pre- and postspillway surface sediment (1293 and 1602 $g_{Surf.sed.}/m^2$, respectively), both pre- and postspillway sedimentary P concentration equaled 3.1 $mg_P/g_{Surf.sed.}$. These values were identical to those measured in the center of the lake (Chapter 2).

Table 5-3. Pre- and postspillway concentrations of sedimentary P in the inorganic, benthos, detrital, and soluble form. Data from the standard model run. Values reported are the average, minimum, and maximum (in g/m^2); time-of-minimum and time-of-maximum (in years after model start).

	Prespillway					Postspillway				
	Avg	Min	Tmin	Max	Tmax	Avg	Min	Tmin	Max	Tmax
Inorganic sed. P	1.17	0.96	15.0	1.46	0.5	1.68	0.97	15.0	2.06	29.5
Benthos sed. P	0.80	0.40	0.0	0.85	5.9	0.92	0.80	16.4	1.08	30.8
Detrital sed. P	1.65	1.37	0.1	1.73	6.0	1.90	1.64	15.6	2.20	30.9
Pore-fluid sed. P	0.37	0.31	15.0	0.47	0.5	0.54	0.31	15.0	0.66	29.5
Total sed. P	4.00	3.76	0.0	4.17	3.5	5.04	3.81	15.0	5.61	29.5

Postspillway *in situ* sedimentation rates were 45, 19, and 10% higher at increasing distances from the dam (Chapter 2). In the model, sedimentation rate increased 27% from 647 to 823 g $m^{-2}y^{-1}$. Hence, *in situ* and simulated rates compared reasonably well.

Both the similarity between measured and modeled changes for sedimentary P and for sedimentation rate, as well as the plausible behavior of the model for water-column P, pore-fluid P, and seston add support to the feedback loop hypothesis. The decrease in organic matter content of surface sediments in the simulation does not correspond to field data. This may, in part, be corrected by adjusting the estimate for the equilibrium constant in the sorption reaction between P in pore-fluid and P in inorganic sediment.

Sensitivity Analysis

The objectives of the sensitivity analysis were: (1) to enhance understanding of model behavior, (2) to test the model for sensitivity to errors in parameter estimates, and (3) to identify sensitive parameters which require further field and/or laboratory study to increase confidence in the model.

All sixty-eight model parameters were individually halved and doubled. Parameters involved with the fractionation of plankton, benthos, and littoral material were changed by increasing and decreasing the largest fraction by 20%, while changing the remaining two fractions proportionally to maintain a total of 100%. The effect of all parameter changes on water depth, seston, water-column SRP, organic matter content of surface sediments, and sedimentation rate is given in Table 5-4. Output characteristics of interest included for both pre- and postspillway conditions: (1) changed averages, minima, and maxima as percent of standard run output, and (2) altered time-of-minimum and time-of-maximum as percent of total simulation time (40 years). The parameters in Table 5-4 were grouped per output variable and were ranked in descending order of activity relative to the percent change in the average value of the standard run output. Parameter activity is defined as the absolute value of the percent change in an output variable when the parameter is halved, plus that when the parameter is doubled. A "+" sign indicates that doubling the parameter increases and halving decreases the value of the output variable. A "-" sign indicates the opposite. Values in italics indicate that both changes of the parameter cause the output to move in the same direction. This occurs when the initial parameter value was optimal. Only changes greater than 25% are reported.

Water depth was relatively insensitive to halving and doubling of individual parameter values (Table 5-4). Only a few parameters produced changes greater than 25% in water

Table 5-4. Results of sensitivity analysis. Values reported under average, minimum, and maximum are the sum of the absolute values of the percentage changes in the output-variable (compared to standard run output) when the parameter is doubled and halved. Values reported under time-of-minimum and time-of-maximum represent the sum of the absolute values of the change compared to standard run output (as percentage of total simulation time) when the parameter is doubled and halved. A + sign indicates that doubling the parameter increases, and halving decreases the value of the output variable. A - sign indicates the opposite. Values in italics indicate that both parameter changes cause the output to move in the same direction. Only changes >25% are reported. See Table 5-1 for parameter description.

OUTPUT VARIABLE	DEPTH (m)									
	PRE-SPILLWAY					POST-SPILLWAY				
Parameter	Avg	Min	Tmin	Max	Tmax	Avg	Min	Tmin	Max	Tmax
Output Standard Run	1.50	0.94	14.7	2.06	0.30	1.59	1.17	39.7	2.01	16.2
AMPNODAM		-85	-32	+41	+27					
AMPDAM							-47		+29	
MAXBENTHICBACT							+31			
BENTHICBACTFRAC							+32			
PBENTHICBACTCONT							+32			
PBENTHICEUKCONT							-32			
Q			+35							
SUBMLITAREA			+35							
N			-32		+27					
BENTHOSSPECREGRT							-32			

Table 5-4--continued.

OUTPUT VARIABLE	SESTON (g m-3)									
	PRE-SPILLWAY					POST-SPILLWAY				
Parameter	Avg	Min	Tmin	Max	Tmax	Avg	Min	Tmin	Max	Tmax
Output Standard Run	+28.8	20.0	0.0	31.3	4.7	34.8	27.4	15.3	38.3	31.8
ALPHA	+174	+43		+192	+37	+264	+230		+292	
BENTHOSSPECREGRT	+32			*90*	+35	+155	+103		+203	+33
PBENTHICEUKCONT	+32			*90*	+35	+155	+103		+203	+33
PBENTHICBACTCONT	-32			*89*	-35	-153	-102		-201	-33
BENTHICBACTFRAC	-32			*88*	-35	-153	-102		-202	-33
MAXBENTHICBACT	-31			*82*	-35	-147	-97		-193	-33
SUBMLITAREA	+116	+29	-36	+109	+27	+134	+138		+132	
SVELP	-54		+36	-78	-33	-132	-98		-164	-33
Q	+91		-36	+92	+35	+130	+124		+131	
PLANKSPECREGRT				*63*	+33	+114	+74		+152	+33
PEUKCONT				*62*	+33	+108	+70		+144	+33
PBACTCONT	-47	-36		-62	-33	-108	-83		-135	-33
SDR	-84			-86	-37	-101	-101		-103	
PLITPLANKTONCONT	+52			+53		+66	+61		+68	
SVELD	-56			-53		-59	-62		-58	
PDETRITUSCONT	-57	-60		-56		-56	-61		-55	
BACTFRAC				*26*	-33	-55	-40		-70	-33
PAUTOCONT	-25			-26	-33	-51	-42		-59	-33
AMPNODAM	-47	-47	+33				-71			
N	-46	-39	+33				-66			
PINORGCONT	-28	-30		-26		-36	-27		-38	
PDELLITPLANKTONCON	+33			+35		+29	+39		+26	
AMPDAM						-27		+30		*42*
KT	-27			-26	+25					
SVELI						-25			-25	
M				+27	-27					
BLIFETM					-30					
KDETSED					-30					
BULKDENS					-27					
KM1					+27					
KR2					+27					

Table 5-4--continued.

OUTPUT VARIABLE	SRP-WATER (g m-3)									
	PRE-SPILLWAY					POST-SPILLWAY				
Parameter	Avg	Min	Tmin	Max	Tmax	Avg	Min	Tmin	Max	Tmax
Output Standard Run	0.016	0.006	0.0	0.022	0.7	0.023	0.012	15.2	0.028	31.7
ALPHA	+764	+69		+945	+37	+1056	+1173		+1255	
BENTHOSSPECREGRT	+566	+58	-37	+794	+37	+1010	+994		+1405	+33
PBENTHICEUKCONT	+565	+58	-37	793	+37	+1008	+992		+1402	+33
BENTHICBACTFRAC	-561	-61	+37	-784	-37	-1000	-985		1389	-33
PBENTHICBACTCONT	-560	-58	+37	-784	-37	-999	-983		-1388	-33
MAXBENTHICBACT	-534	-61	+38	-742	-37	-951	-938		-1312	-33
PLANKSPECREGRT	+430	+52	-38	+583	+37	+757	+755		+1010	+33
PEUKCONT	+413	+51	-38	+557	+37	+726	+725		+964	+33
SVELP	-367	-48	+38	-482	-37	-634	-637		-828	-33
PBACTCONT	-301	-42	+35	-379	-37	-519	-514		-673	-33
BACTFRAC	-197		+35	-218	-35	-320	-320		-399	-33
PAUTOCONT	-151		+35	-142	-35	-221	-228		-258	-33
KR1	-106			-136		-89	-103		-85	
SUBMLITAREA	+88			+57		+99	+116		+99	-30
KM1	+79			+50	+35	+98	+119		+106	
Q	+69			+44	+35	+96	+105		+99	
BULKDENS	-65			-37	-33	-72	-96		-76	
KDETSED	-54			-32	-35	-69	-82		-83	-30
BLIFETM	-54			-33	-35	-69	-82		-83	-28
PLIFETM	+45					+63	+59		+71	
BEDDENS	+36					+58	+57		+63	+28
V1	-55			-81		-43	-45		-49	
MAXBENTHICAUTO	-40					-53	-53		-59	
PLITINORGCONT	+35					+52	+48		+58	
SVELI	-50			-54		-51	-57		-43	
KR2	+31			*36*	+35	+51	+72		+51	
RISEDAM						+50			+64	
AMPNODAM	-50			+88	-25		-104			
N	-45		+35			-30	-78			
PDETSEDCONT	-40					-34	-58		-34	
SVELD						-38	-34		-44	
MAXPAR	-27					-37	-35		-42	
MAXPLANKAUTO	-27					-37	-35		-42	
OPTPAR	+27					+37	+35		+42	
SDR	*26*				*62*	*36*	*56*		*48*	
AMPDAM						-35				-30
PLITPLANKTONCONT						+26	+30			
PDELLITINORGCONT							+30			
PINORGSEDCONT							-28			

Table 5-4--continued.

OUTPUT VARIABLE	SED.ORG. MATTER (%)									
	PRE-SPILLWAY					POST-SPILLWAY				
Parameter	Avg	Min	Tmin	Max	Tmax	Avg	Min	Tmin	Max	Tmax
Output Standard Run	0.70	0.61	0.5	0.74	15.0	0.64	0.61	31.5	0.74	15.0
BENTHICBACTFRAC	+131	+103	-37	+44	*73*	+152	+159	-59	+129	*66*
PBENTHICEUKCONT	-131	-103	+37	-44	*73*	-151	-158	+59	-128	*66*
BENTHOSSPECREGRT	-131	-103	+37	-44	*73*	-151	-158	+59	-128	*66*
MAXBENTHICBACT	+129	+102	-37	+44	*73*	+150	+159	-59	+127	*66*
PBENTHICBACTCONT	+130	+101	-37	+43	*73*	+150	+158	-59	+128	*66*
PLANKSPECREGRT	-108	-90	+36	-39	*68*	-136	-143	+58	-114	
PEUKCONT	-106	-88	+36	-39	*68*	-134	-141	+58	-112	
ALPHA	-104	-85	+36	-39	*70*	-131	-138	+58	-109	
SVELP	+101	+83	-36	+39	*68*	+129	+137	-58	+108	
PBACTCONT	+88	+72	-36	+37	*68*	+121	+129	-58	+97	
BACTFRAC	+65	+51	-36	+33	+38	+96	+104	-55	+74	
PAUTOCONT	+54	+35	-36	+33	+38	+80	+87	-53	+60	
PINORGSEDCONT	+49	+54		+45		+58	+62		+45	
SDR	-37		+25		-37	-49	-54	-40	-41	
KR2	-37		+34	-28	-37	-49	-52	+25	-41	
KM1	-31		+33	-27	-37	-40	-42		-37	
BULKDENS	+26		-25	+25	+37	+32	+32		+31	
PDETSEDCONT		-45				-30	-34			
PLIFETM						-29	-33			
KDETSED			-35		+35			-38		
BLIFETM			-35		+35			-35		
BEDDENS							-27	+43		
SUBMLITAREA								-40		
Q								-40		
PLITPLANKTONCONT								-35		
AMPNODAM								+30		
MAXBENTHICAUTO							+27			
AMPDAM								-27		
RISEDAM								+25		

Table 5-4--continued.

OUTPUT VARIABLE	SEDIMENTATION RATE (g m−2 y−1)									
	PRE−SPILLWAY					POST−SPILLWAY				
Parameter	Avg	Min	Tmin	Max	Tmax	Avg	Min	Tmin	Max	Tmax
Output Standard Run	647	595	0.1	669	4.4	823	629	15.0	879	31.4
PBENTHICBACTCONT	-127	-26	+38	-197	-36	-314	-241		-389	-34
BENTHOSSPECREGRT	+124		-38	+198	+36	+314	+240		+390	+34
PBENTHICEUKCONT	+124	*25*	-38	+197	+36	+313	+240		+389	+34
BENTHICBACTFRAC	-124	*25*	+38	-195	-37	-312	-239		-387	-34
MAXBENTHICBACT	-118		-38	-185	-37	-297	-227		-370	-34
PLANKSPECREGRT	+93		-37	+147	+38	+239	+182		+297	+34
PEUKCONT	+89		-37	+141	+38	+230	+175		+285	+34
SVELP	-77		+37	-120	-34	-202	-152		-250	-34
PBACTCONT	-66		+37	-96	-34	-168	-126		-208	-34
ALPHA	+84			+100	+36	+160	+136		+184	
SUBMLITAREA	+109	+45	-37	+83	+26	+124	+133		+122	
Q	+85	+38	-37	+74	+36	+121	+119		+120	
BACTFRAC	-44		+37	-57	-33	-106	-80		-129	-32
PDETSEDCONT	-101	-95	+37	-95		-94	-113		-91	
SDR	+75	+104	+37	+117	-36	+73	+64		+77	
PAUTOCONT	-34		+37	-37	-33	-74	-59		-86	-32
PINORGSEDCONT	-53	-55		-54		-64	-50		-68	
PLITPLANKTONCONT	+46		-37	+35		+57	+54		+57	
AMPNODAM	-49	-59	+35		-29		-75			
KDETSED	-39	-27	+37		-36	-46	-49		-49	
KR2	+28		-37		+36	+46	+40		+49	
BLIFETM	-38	-26	+37		-36	-46	-48		-47	
N	-45	-53	+35		-26		-58			
KM1			-37		+30	+34	+32		+36	
BEDDENS			-37			+31			+33	
AMPDAM						-30		+35	-30	-35
PDELLITPLANKTONCON	+29		-37				+35			
PLITINORGCONT			-37			+28			+30	
KT	-26									
BULKDENS			+37		-28					
B3			-37							
MAXBENTHICAUTO			+37							
MAXPLANKAUTO			+37							
MAXPAR			+37							
PLIFETM			-37							
BWDNODAM			+37							
RISEDAM									+32	
M			-27							

depth. Halving or doubling the amplitude of water-level fluctuations resulted, of course, in considerable changes in the minimum and maximum water depths. With the exception of the effect of ALPHA on model behavior (discussed below), the remaining four output variables were all most-strongly affected by changes in the following parameters: (1) the regeneration of P by eukaryotic heterotroph benthos, (2) the P content of benthic bacteria, (3) the P content of eukaryotic heterotrophs, (4) the bacterial fraction of benthos, and (5) the maximum P-uptake rate for benthic bacteria. The feedback loops in which these parameters are involved occur in the surface sediments and affect the rates of P exchange between bacteria and benthic heterotrophs with the interstitial pore-fluid loops 21, 22, and 28; Figure 5-5). Research focused on improving the estimates of these active parameters may enhance confidence in the model. The rate of P uptake by benthic bacteria, in particular, has no strong foundation in the literature. Changes in parameters involved in P exchanges with detritus in the form of rates of death and decomposition have a much smaller impact on model behavior.

ALPHA, a model coefficient determining the specific rate of resuspension of surface sediments (loops 7-10; Figure 5-3), is the most active parameter for seston and water-column SRP concentrations. Its effects on organic matter content of the sediments and sedimentation rate, however, are considerably lower than the above-discussed coefficients. Similarly, changes in the settling velocities of inorganic and detrital suspended matter are of intermediate significance to seston and water-column SRP concentrations, but have no noticeable influence on sediment organic matter content or sedimentation rate. Therefore, the modeled rate of sedimentation (i.e. the rate at which surface sediments are converted to permanent sediments) is not as much a function of the net difference between settling and resuspension, but more a function of benthic processes acting on sediments once they have settled to the bottom.

To test whether the five statistics (used in Table 5-4) were adequate in describing changes in model behavior due to halving and doubling of parameter values, dynamics for the two most active parameters were analyzed. Doubling ALPHA (Figure 5-8) and halving PBENTHICBACTCONT (Figure 5-9) showed that no major pattern changes (additional inflection points in the plots) occurred that were different than those from the standard run output (Figure 5-7). The relative stability of sedimentary organic matter content in the standard run was repeated after the model output passed through an starting transient caused by a high initial value. Pore-fluid SRP is absent from these two graphs (compared with the standard run) because it was

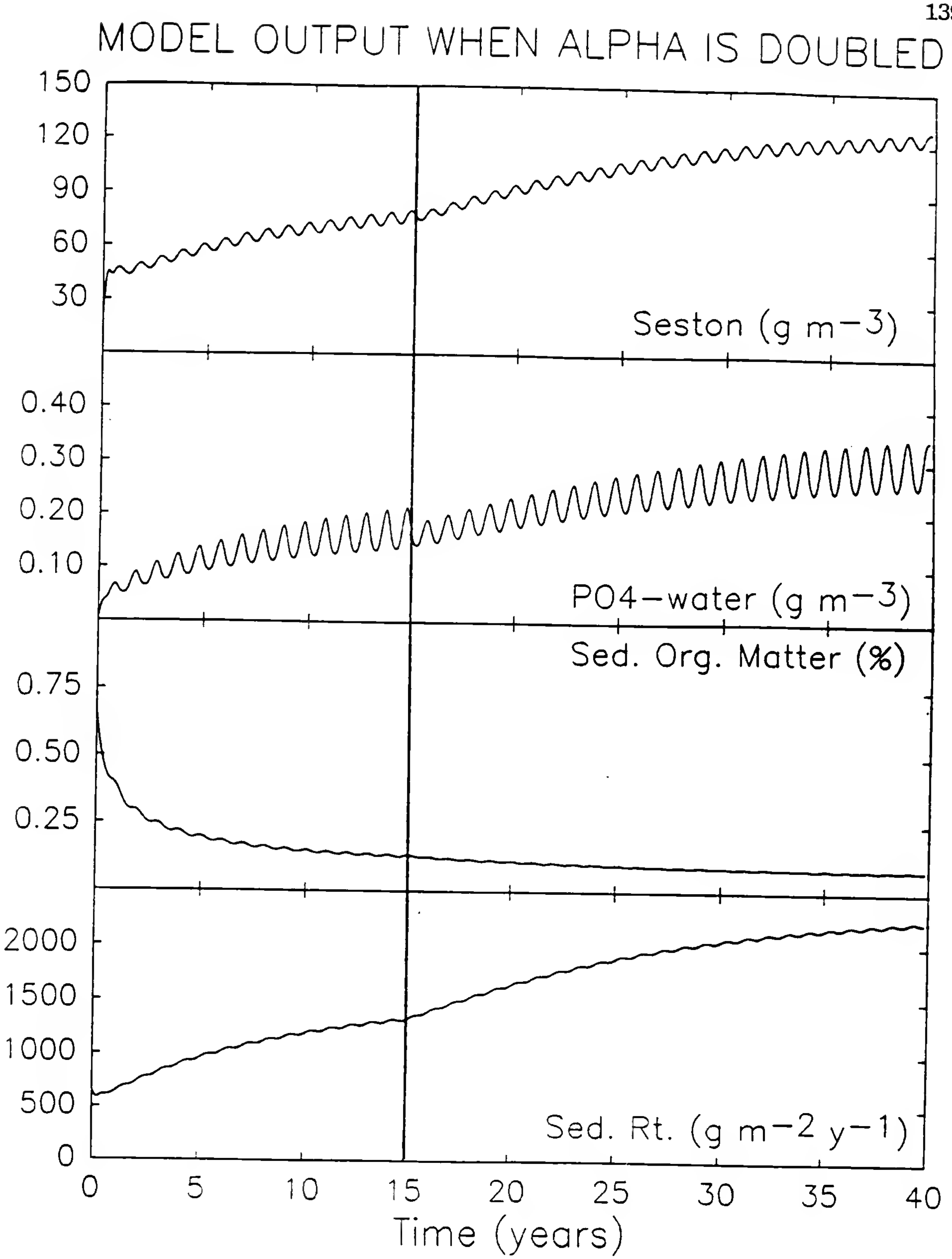


Figure 5-8. Model output when ALPHA is doubled. Spillway installation initiated at 15 years.

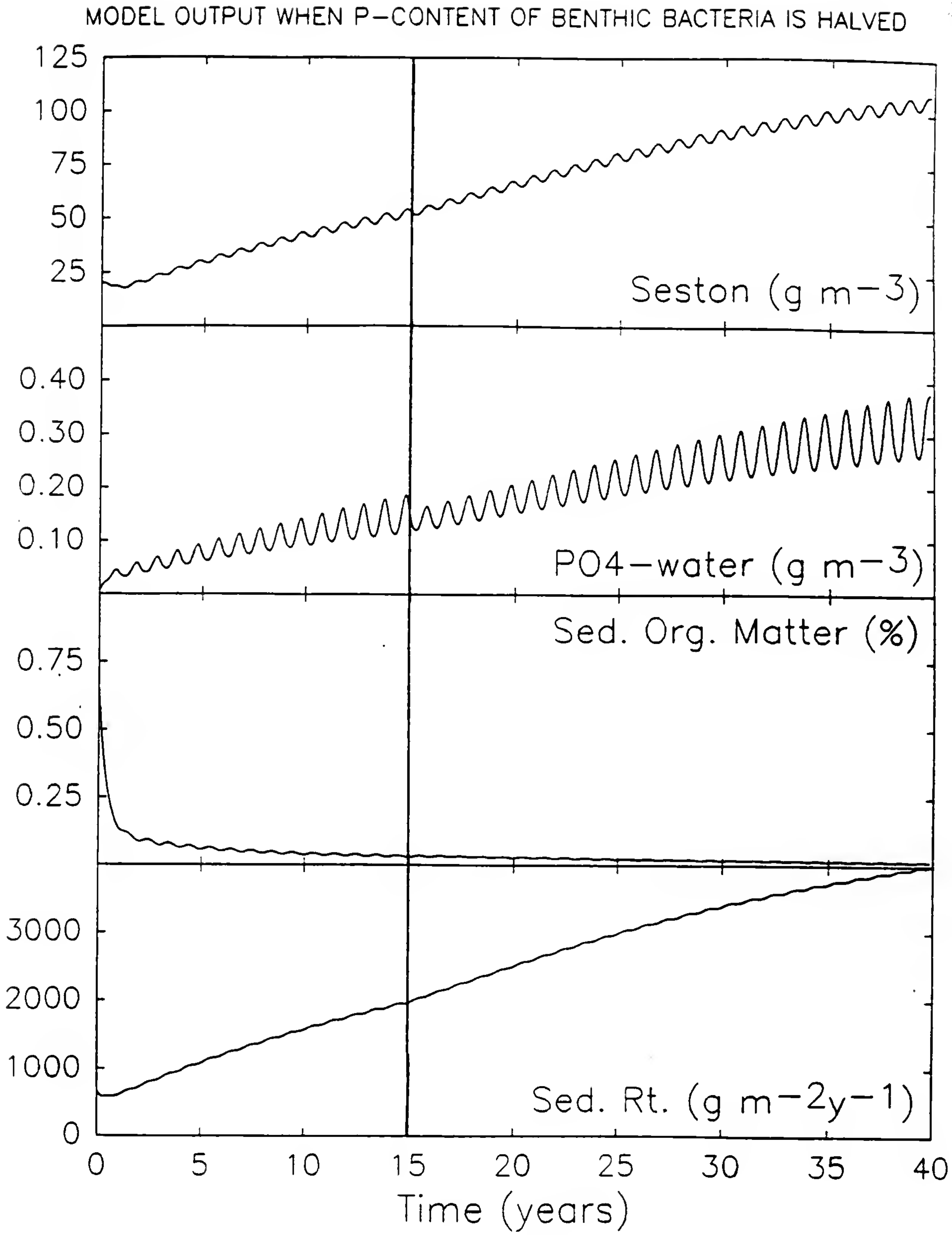


Figure 5-9. Model output when the P content of benthic bacteria is halved. Spillway installation initiated at 15 years.

not included in the sensitivity analysis. Water-column depth, which was part of the analysis, is not shown because it was not significantly affected by the two parameters involved in this test.

The plausibility of the model response to a change in a parameter may be analyzed by following this change through the equation structure. For example, doubling ALPHA (Figure 5-8) doubles the specific rate of resuspension. This results in an increase in seston and water-column SRP (hence the "+" signs for the influence of ALPHA on these two model variables in Table 5-4). The magnitude of this increase depends on water depth. More seston, in turn, produces a higher rate of settling. This rate, however, is not the same for all fractions of seston. Inorganic seston settles faster than planktonic seston by an order of magnitude. This produces a rapid decline in organic matter content of the surface sediments (hence the "-" sign for the influence of ALPHA on SEDORGMAT in Table 5-4). Increased settling leads to more surface sediments. This positive resuspension-settling loop is limited by the effect of negative loops between resuspension and surface sediments and between seston and settling (see also Figure 5-3). The increase in water-column SRP, resulting from doubling of ALPHA, may elevate primary production. This produces more seston and increased settling, which leads to more surface sediments and higher rates of sedimentation.

Again, model behavior is plausible as shown by the "+" signs for the influence of ALPHA on sedimentation rate (Table 5-4).

Shifts in the time of peak occurrences were similar among changes in different parameters. An example may clarify this. While the values of prespillway minima were generally stable, their occurrence was often 35 to 38% later than in the standard run. This change always occurred when either halving or doubling the parameter value changed the model variable from gradually increasing to steady state (i.e. time-of-minimum is zero) to gradually decreasing to steady state (i.e. time-of-minimum is 15 years or 37% of simulated time later). The variability of time-of-minimum values between 33 and 38% was caused by the difference in phase of the sinusoid time pattern at t=15 years. This likely resulted from inherent inaccuracies of calculations performed on a digital computer which are compounded during the 80,000 iterations performed in one model run.

In general, the parameters associated with activity in the littoral zone were of intermediate or low activity. Changes in the efficiency of the wetland fringe to trap particulate matter and P from runoff from the watershed had no noticeable effect on model output. Since this trapping efficiency was proportional to the extent of the wetland fringe, the latter had only a minor impact on P and seston loading of the profundal zone in this model. Although this loading increased after spillway installation, it remained generally less than 1% of the total input to the profundal water-column dominated by internal loading through resuspension of the surface sediments and pore-fluid.

The effect of light on model output was also of low activity. A doubling of MAXPAR (maximum available photosynthetically active radiation at average lake depth) doubles PAR. The effect of this on P-uptake rate is, however, reduced by the way in which PAR and its effect on specific P-uptake rate (PAREFF) relate (parabolic, concave to the x-axis; see Figure 5-4b) and by the manner in which PAREFF affects the specific P-uptake rate (as the square root of the product of PAREFF and PEFF, the effect of SRP on the specific P-uptake rate).

Individual changes in parameters involved in the effect of the dam on amplitude of water-level fluctuation (AMP), concentration of seston in the outflow (BWD), and hydraulic residence time (HYDRESTM) were also of low activity.

<u>Analysis of Alternative Management Strategies</u>

One of the model objectives was to evaluate the consequences of alternative management actions in Newnan's Lake. The dam may accelerate sediment and detritus accumulation by altering biological processes (primary production), changing sediment dynamics (settling and resuspension), and by hydrological modifications (increased water residence time) (see Figure 5-1). Each of these can be evaluated separately with the model by manipulating the set of parameters involved in each pathway. By ranking the relative significance of each pathway on model output, risks and benefits of management strategies to intended uses of the lake may be compared. The differences in model output (compared to the standard run) for these three management scenarios are given in Table 5-5.

Table 5-5. Change in model output variables (as percent of standard run output) for simulations addressing each modeled impact of the dam separately and comparing this with results from a simulation with no dam installed. See text for a description of the simulations. The statistic reported is the average value of the model output variable postspillway.

Output variable	Seston	Water Depth	Water-Column SRP	Sed.org. matter	Sediment. rate
Littoral area	-6.3	0.3	-8.5	0.1	-6.3
Resus-pension	1.2	-0.1	2.4	-1.6	1.1
Water resi-dence time	-10.4	-7.4	-31.4	13.2	-16.1
$\sum$ Impacts	-15.5	-7.2	-37.5	11.7	-21.3
Model run without dam	-19.2	-7.0	-39.2	14.5	-27.2

In the "littoral area" run, the wetland fringe around the lake was kept at 20% of lake surface area, while other impacts of the spillway were maintained. Such a scenario may be accomplished by hand-planting of vegetation in the littoral area of the lake. This labor-intensive program has been applied to large areas of the Newnan's Lake shoreline since 1990 (Krummrich, Florida Game and Fresh Water Fish Commission, pers. comm.). Its effect on model output is minor, although it may assist in creating refuge, spawning, nursery, and feeding grounds for many species of fish (Durant 1980; Schramm et al. 1983; Conrow 1984).

In the "resuspension" run, the rate of resuspension of surface sediments after spillway installation was maintained at prespillway levels. Its effect on model output was also minor (Table 5-5). In other words, the reduction in sediment resuspension produced by water-level stabilization and the effect of this on sediment flushing was of minor importance to the rate of sediment and detritus accumulation in the lake. In a separate run, the model also predicted that modifying the dam to allow bottom-water drainage will have little effect. Maintaining the bottom-water drainage at prespillway levels reduced sedimentation rate by only 3%.

The impact of the dam on lake hydrology is of greater significance to the model behavior (Table 5-5). Maintaining the water residence time at prespillway levels and eliminating the rise in water level after spillway installation reduced sedimentation rate by 16% and water-column SRP by 31%. Management strategies designed to reduce water residence time, such as gravity drawdowns, appear most effective in reducing sedimentation rate. A more extreme water-level reduction, such as may be accomplished under a mechanical pumpdown, was simulated by lowering the water level to the historic low of 19.65 m (msl) for a period of 6 months. Shallow water depth caused increased resuspension of bottom material, which produced 15-20% higher levels of seston and water-column SRP during the period of pumpdown. Increased resuspension also lowered the amount of surface sediments by 8% during 3-4 years.

The combined effects of dam installation on model output are illustrated by a model run without the dam. Without the dam, the average rate of sedimentation was 27% lower, and water-column SRP and seston were reduced by 39 and 19%, respectively (Table 5-5). It is interesting to note the difference between the sum of the separate effects of the spillway installation and the combined effects in the model run without the dam (Table 5-5). Some non-linearity appears to exist resulting from an interaction among the ways in which the dam affects sediment and P dynamics in the lake. Such an interaction occurs, for instance, between the ways the dam may influence phytoplankton production (see Figure 5-1). After spillway installation, production may be enhanced by both increased light availability due to reduced sediment resuspension and by increased external nutrient loading caused by a reduction in littoral vegetated fringe.

The spillway was built in 1967 with the management intent of raising the lake stage and eliminating extreme low-water levels (Smith and McGriff 1958). Hydrologic records (Adkins 1991) indicated that its benefit to lake access and use for outboard, recreational boats has been minor. The model demonstrated a 7% increase in average water depth postspillway. Much of this increase has been eliminated after 25 years by higher rates of sedimentation. Increased seston concentrations may decrease the aesthetic value of the lake and oppose lake management goals to promote the lake for water recreation. A 39% increase in simulated water-column SRP may increase plant and/or algal biomass and further interfere with recreation and navigation. While the extent of the vegetated littoral zone and its ability to filter particulate matter and P from runoff from the surrounding watershed likely has been reduced by water-level stabilization, this reduction has not impacted sediment accumulation rates in the model.

Short-term removal of the dam produced plausible results in the model (Figure 5-10). Conditions in this simulation were set to mimic the short-term drawdown and drought of the spring of 1989, documented in Chapters 3 and 4. This was done by simulating removal of the spillway at t=30 years and lowering the water level to 19.85 m msl for 3 months. Low water levels increased resuspension of surface sediments and pore-fluid SRP. More suspended

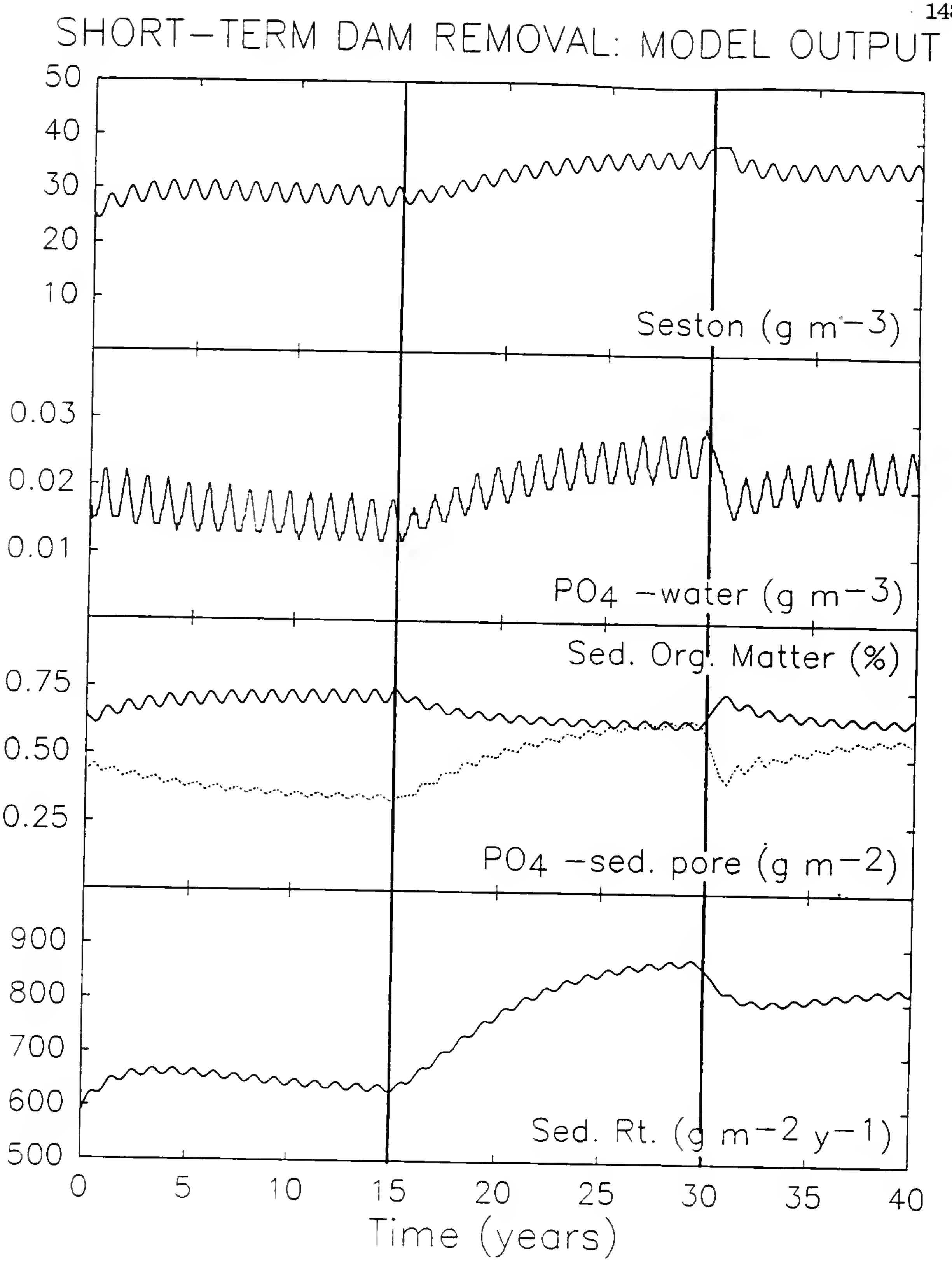


Figure 5-10. Model output with spillway installation at 15 years and removal of the spillway at 30 years for a period of 3 months.

matter in the water column depleted water-column SRP, and produced, after a settling delay, a higher organic matter content in the surface sediments. While the plausibility of this generated time pattern provides support for the feedback hypothesis, a more rigorous validation is desirable. An example of such a test is to measure seston or water-column SRP in the field before, during, and after the short-term drawdown and compare these measurements with the model output.

The current long-term dam removal (initiated in 1991 based on the results of my field studies) provides future opportunities for model validation. Monthly water quality sampling is now carried out in Newnan's Lake (Ware, St Johns River Water Management District, pers. comm.) and may be helpful in this respect. The effect of long-term spillway removal on model behavior is shown in Figure (5-11). Removal was simulated by resetting the parameters directly involved with the dam to prespillway conditions at t=30 years. The model output variables showed a gradual return to prespillway levels over a time period in excess of 10 years.

Summary and Conclusions

The model has synthesized the best, current understanding of the impact of water-level stabilization in Newnan's Lake. This was done in the form of a feedback loop hypothesis based on field data, literature, opinion,

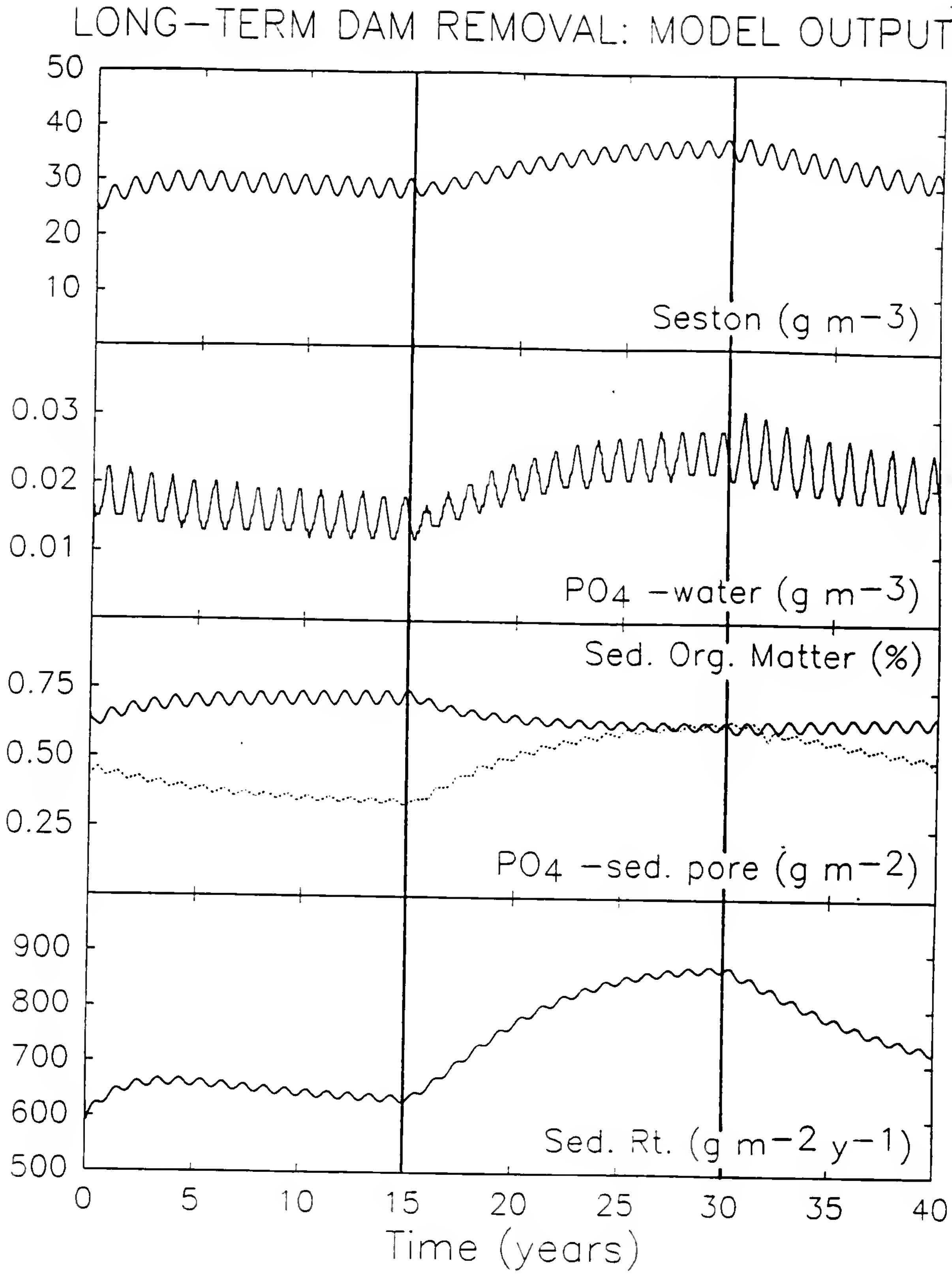


Figure 5-11. Model output with spillway installation at 15 years and removal of the spillway at 30 years.

and estimates. Since the general behavior of the model was quite similar to available real-system dynamics, and since the simple response pattern of the model was not altered by systematic parameter changes, support has been gained for this feedback loop hypothesis as an adequate explanation of sediment and P dynamics in this aquatic ecosystem.

The model is most sensitive to changes in feedback loops involved in sediment resuspension and in actions of bacteria and benthic heterotrophs on the production and movement of P in the surface sediments. Hence, parameters affecting P flows between benthos and sedimentary pore-fluid and those affecting wind-wave induced resuspension require additional investigation to enhance confidence in the model. Field measurements of these processes are scarce. Future work should focus on (1) determining the flow of (labeled) P in surface sediments and (2) quantifying the relationship between wind-wave action, water depth and resuspension of bottom material.

Each effect of the spillway separately had minor implications for the behavior of the model. In combination, however, they resulted in elevated seston, water-column SRP and sedimentation rate. Over a period of 25 years, the intended benefit of the dam (lake access, navigation) has been eliminated and replaced with common lake management "costs" such as increased water-column turbidity, primary production and material deposition in the lake. Spillway removal, recently initiated, provides an excellent opportunity to test further whether the major causal forces between the spillway and sediment-P dynamics in Newnan's Lake have been identified in this model.

CHAPTER 6
SUMMARY AND DISCUSSION

The long-term significance of a spillway in the outlet of Newnan's Lake for the transfer of bulk sediment, organic matter, and nutrients between water and sediment was evaluated using a paleolimnological approach. The effect of a short-term removal of the spillway on the transfer of this material out of the lake (e.g. flushing, oxidation) or within the system (e.g. erosion, redistribution) was determined in a number of field and laboratory experiments. The findings of these studies were integrated with independently derived information from literature and theory in a feedback dynamics model, and their implications for lake management were tested in a series of simulations.

Paleolimnological analysis of the sedimentary record in Newnan's Lake allowed interpretations about past conditions of the ecosystem. The sedimentary signal at the time of spillway installation and the agreement between ^{210}Pb and ^{137}Cs derived dates were remarkable, particularly since the shallow depth of the lake and the flocculent nature of the sediments likely produced a more disturbed stratigraphy than that found in the traditional paleolimnology of deeper lakes (cf. Shapiro et al. 1971; Robbins and Edgington 1975; Pilskaln and Johnson 1991). This corresponds with earlier, promising paleolimnological data from other shallow productive lakes in Florida (Brenner and Binford 1986; Deevey et al. 1986) and is encouraging for paleolimnological work in such systems.

The finding in this study that net accumulation rate of bulk sediment increased by 45% since spillway construction, while rates for organic matter and nutrients tripled compared to prespillway conditions, has implications for lake depth and productivity. More sediment deposition increases the rate of filling of the lake. A higher nutrient content of the surficial sediments will enhance internal loading of nutrients through mixing of this material in the water column. As such, the spillway may accelerate the successional development of the lake towards a freshwater marsh. Further paleolimnological work may supply additional lines of evidence to support this conclusion. For instance, changes in lake productivity may be reflected in relative abundance of subfossil _Chironomus_ sp. and _Tanytarsus_ sp. in the sediment profile. These highly ubiquitous midges leave identifiable morphological remains in lake sediments and _Chironomus_ sp. is generally considered to be indicative of a higher trophic state (Crisman 1978). Alternatively, the relative abundance of chydorid Cladocera remains in the sediments may illustrate historical changes in the lake's littoral productivity. Chydoridae are generally non-planktonic and occur in association with littoral and benthic substrate (Goulden 1969).

The short-term drawdown demonstrated the improbability of lowering the water level in the lake by a gravity drawdown to a stage less than that accomplished (19.70 m MSL). Small elevation gradients in the lake basin and the gradual build-up of obstructions in the lake outlet upstream and downstream from the spillway since its construction (Crider 1972; personal observation) likely prevented a more dramatic drawdown of water level. However, dredging of the lake outlet to enhance outflow conditions and increase the amplitude of a drawdown may not be necessary. Periodic removal of the spillway will likely scour the outflow to accommodate larger discharges and return this channel to its historic configuration. During this drawdown, the water level was barely lakeward of the cypress tree fringe and drying and consolidation of lake bottom was limited to a narrow stretch of littoral zone. In other Florida lakes, significant consolidation only occurred when substrate was exposed for a minimum of several months and water levels were significantly below the level of the unconsolidated sediments (Greening and Doyon 1990). A shorter exposure time may result in the formation of a "cap" of dried organic material over unconsolidated wet sediments. This cap may then become dislodged and float to the water surface during and after refill of the lake.

Flushing rates of particulate organic matter and nutrients from the lake during the drawdown were low. This removal, however, was accomplished at low cost since drawdowns are unquestionably among the least expensive lake management techniques (Cooke et al. 1986). Drawdowns may be an appropriate management strategy in a shallow, exposed lake such as Newnan's Lake, where downstream areas are able to absorb particulate matter loadings during the period of flushing.

Maximum flushing of organic matter and nutrients from Newnan's Lake by temporary removal of the dam occurred at a high lake stage and during periods of high average wind speed. Maximum monthly lake stage in Newnan's Lake occurs either from February to April or from July through September (Adkins 1991). No wind data were analyzed for north-central Florida, but data for Lake Okeechobee indicated that maximum wind velocities between October and May were approximately 30 percent greater than between June and September (Maceina and Soballe 1990). Assuming that these data can be applied to Newnan's Lake, it appears that short-term drawdowns during February-April have the highest probability of leading to improved lake conditions. Drawdowns during this time period, however, are not optimal from a fisheries perspective. Fish generally spawn in February in Florida and need nursery areas in spring and early summer. Frequent drawdowns (i.e. every year) during February-April are therefore not recommended if the lake is managed for sportfishing. Spillway closure in May will allow re-filling of the lake by convective summer rains. In order to more firmly establish a recommended time span for board removal, research should focus on (1) analysis of average hourly wind speed throughout the year for the Newnan's Lake watershed, and (2) hydrologic modeling to evaluate the refill potential of the lake following a February-April drawdown compared to a July-September drawdown.

Erosion of flocculent littoral substrate was recorded during the drawdown. Although variability was high. an average of 6.08 g $m^{-2}day^{-1}$ of organic matter (0.58 g_{TKN} $m^{-2}day^{-1}$ and 0.03 g_{TP} $m^{-2}day^{-1}$) was removed from the littoral zone. Sediment depth to sand decreased by an average of 42% after drawdown, and the bulk density of the remaining substrate increased by an average of 250%. Removal of soft substrate from the littoral zone may promote rooted plant growth which can serve as nursery area and refuge for fish. In addition, such removal may improve littoral habitat for future sportfish spawning.

The quantitative record of the transfer of this material to deeper areas of the lake was not clear. The small drop in lake level during the drawdown did not create large areas of erosion in the littoral zone. This reduced the potential redistribution of material in the lake. In addition, the use of field markers would have eliminated error due to imprecise matching of pre- and postdrawdown stations. These markers (e.g. buoys, flags) were not used in the study because of safety concerns for people using the lake for recreation and sportfishing.

A sediment and phosphorus model was developed to synthesize the best current understanding of the effects of the spillway on Newnan's Lake. It combined the results from laboratory and field study with information from literature, opinion, and, where necessary, guesses. Conclusions from the model were given in the previous chapter. Here, the emphasis is on an evaluation of the interaction between the experimental components of this study and the model. Results from field and laboratory work assisted in model development and analysis, especially in quantifying feedback relationships, evaluating model behavior during the sensitivity analysis, and field testing the model. Conversely, the model was used to improve the understanding of the system and to identify specific research areas needing further study. In this case, model analysis suggested research needs in quantifying the relationship between wind-wave action, water depth, and resuspension of bottom material, as well as measuring the flow of phosphorus in surface sediments between benthos and interstitial pore-fluid. The actions of bacteria and benthic heterotrophs showed a considerable impact on the production and movement of phosphorus and, thus, on the productivity of the overlying water. This corresponds with the key areas of research needs identified in a comparative study of a large number of lake ecosystem models (Scavia 1979).

Modeling has advanced the understanding of this aquatic system by: (1) pointing-out errors in thinking when the model did not adequately reproduce what was known to be true about the behavior of the real system, (2) requiring explicit statements about separate processes and the way in which they interact, (3) encouraging continued feedback between the model and the real system, and (4) by forcing the integration of theory from disciplines (e.g. hydrology, rheology, ecology, lake management) not routinely combined.

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APPENDIX A
LEAD-210 (^{210}PB) AND CESIUM-137 (^{137}CS) DATING OF SEDIMENTS USING LOW-ENERGY, INTRINSIC-GERMANIUM, γ-RAY SPECTROSCOPY

Application

Direct γ-ray assay is used to record changing concentrations of ^{210}Pb (unsupported and supported) and ^{137}Cs in sediment profiles. This forms a reliable and precise basis for age/depth and dry-sedimentation-rate calculations.

Counting system

Location

The system used for spectral analysis is located at the University of Florida's Department of Environmental Engineering Sciences' Low-Background Counting Room. This room was designed to reduce the level of background radiation interference during sample counting.

Detector specifications

Manufacturer	Princeton Gamma Tech, Inc.
Model	Intrinsic Germanium Well 12225-14
Preamplifier	RG11B/C
Crystal geometry	P-Type Well

Detector Bias	3000 V
Crystal Size	61 mm (ø), 60 mm (length),
	139 cm^3 (active volume)
Well Size	14 mm (ø), 40 mm (length)
Well Material	Vespel (0.51 mm)
Bias Polarity	Positive
Efficiency (counts/γ)	0.57 (@ 46.5 keV)
	0.27 (@ 295.2 keV)
	0.22 (@ 351.9 keV)
	0.08 (@ 609.3 keV)
	0.07 (@ 661.6 keV)
Standard Peak/Compton	24:1 - 90:1
FWTM/FWHM	1.9 - 2.0
Lowest Detectable Energy	9 keV

Background Suppression

Background suppression is accomplished with a 10.1 cm thick lead shield (low activity). In addition, evaporated liquid nitrogen is piped into the detector shield to create excess pressure and displace ambient air during operation.

Operating Conditions

The detector operates at a temperature near that of liquid nitrogen (-196°C), which is provided by a high vacuum cryostat-dewar system (capacity 30 l). Liquid N_2 use is approximately 2 l/day.

Electronics

System electronics include:

Amplifier	EG & G Ortec, 672
Bias Supply	Princeton Gamma Tech, 5 kV, 315A
Power Supply	EG & G Ortec, 402M
MCB	EG & G Ortec, ACE 4K PC Card

Computer and Software

A computer with multi-channel analyzer ("Maestro" ADCAM 100, EG & G Ortec, 0.5 keV/channel) and mathematical spreadsheet (Quattro, Borland Inc.) are used for data conversion and analysis.

Procedure

1. Sediment cores , kept at 4°C, are sectioned in 1 cm intervals within 48 hours after collection. Separate sections are stored in plastic at 4°C.
2. Bulk density (g/cm^3) is determined by drying 1 cm^3 subsamples at 95°C for 24 hours followed by cooling and weighing on an analytical balance.
3. Samples for isotope analysis are dried at 95°C for 24 hours, pulverized by mortar and pestle, re-dried (95°C, 1 hour), weighed, and placed in small, low-density polypropylene tubes (capacity 4 ml). The sample volume and density should correspond to the volume and density of the standard to insure the same counting efficiencies for both. The samples and standard are covered with an identical seal

(epoxy) and archived for 14 days to equilibrate radon (^{222}Rn) with radium (^{226}Ra).

4. Counting times vary from 7 to 14 hours depending on the weight of the sample; low weight samples need longer counting times to reduce error. A blank is counted for every three samples to determine background levels of radiation. Standards are run with the same frequency to track efficiency (counts/γ) and calculate a ^{226}Ra conversion factor (pCi/cps).

5. Sample spectra are analyzed for activity in the 46.5 keV (^{210}Pb) and 662 keV (^{137}Cs) peaks. Activities at 295 keV (^{214}Pb), 352 keV (^{214}Pb), and 609 keV (^{214}Bi) representing uranium-series peaks are used to compute supported levels of ^{210}Pb.

Calculations and Accuracy

Radio-nuclide Concentrations

The concentration (pCi/g) for each γ-energy of interest is determined with

$$C = \frac{P_c - B_c}{E \times I \times t \times g \times 0.037} \tag{A-1}$$

where P_c = Peak counts

B_c = Background counts

E = Efficiency (counts/γ)

I = Intensity (γ/disintegration)

t = Count time (sec)

g = Sample weight (gram)

and 0.037= Conversion factor from disintegrations/sec to pCi.

Counting efficiency for each peak is given by

$$E = \frac{P_c - B_c}{A \times I \times t \times 0.037} \quad \text{(A-2)}$$

where A = Activity of the standard (pCi)

The total ^{210}Pb concentration is calculated using Equation (A-1) applied to the 46.52 keV γ peak. The supported ^{210}Pb concentration is determined by Equation (A-1) applied to the 295, 352 and 609 keV peaks, and averaging. The unsupported ^{210}Pb concentration is then calculated by

$$\text{Unsupported } ^{210}Pb = \text{Total } ^{210}Pb - \text{Supported } ^{210}Pb \quad \text{(A-3)}$$

^{137}Cs concentration is determined by applying Equation (A-1) to the 662 keV peak.

Background

The background characteristics of the system for selected isotopes were optimized by a series (N=13) of blank counts during a 10 week period. Counts per minute (over 840 minutes) were as follows:

^{210}Pb @ 46.5 keV	^{214}Pb @ 295 keV	^{214}Pb @ 352 keV	^{214}Bi @ 609 keV	^{137}Cs @ 662 keV
0.02 ± 0.02	0.01 ± 0.01	0.04 ± 0.02	0.02 ± 0.02	0.04 ± 0.02

These levels are sufficiently low and constant to achieve acceptable precision and counting times. Background counts in the energy ranges of interest need to be monitored after every 3-4 sample counts.

Error Prediction

Analytical error prediction follows nuclear statistics (Knoll 1979) assuming a Poisson distribution for the recorded counts. Uncertainty (pCi/g) depends linearly on counting time and sample weight. Error bars (± 1σ) for Total ^{210}Pb were 0.51 pCi/g (6.3%) for a very small sample (0.723 g) with an activity of 8.14 pCi/g and a 14 hr. count time. A 2.087 g sample with an activity of 6.41 pCi/g counted for 7 hr. produced 0.53 pCi/g (8%) error bars. These errors represent only internal uncertainty (i.e. associated with the accuracy of γ-ray detection).

Calculation of ^{210}Pb Dates

Calculation of ^{210}Pb dates follows the Constant Rate of Supply model (Goldberg 1963; Appleby and Oldfield 1978;

Robbins 1978). As such, the cumulative residual unsupported ^{210}Pb, A_t, beneath sediments of age t varies according to

$$A_t = A_0 e^{-kt} \tag{A-4}$$

where A_0 = Total residual unsupported ^{210}Pb (pCi/cm^2)

k = ^{210}Pb radioactive decay constant

A_t and A_o are calculated by direct numerical integration of the ^{210}Pb profile. The age of sediments of depth x is then given by:

$$t = \frac{1}{k} \ln \frac{A_0}{A_t} \tag{A-5}$$

The sedimentation rate (r) can then be calculated directly (Appleby & Oldfield 1978):

$$r = \frac{k A_t}{C} \tag{A-6}$$

where C = Concentration of unsupported ^{210}Pb (pCi/g) in layer of interest.

Cesium-137

First occurrence of ^{137}Cs in the profile generally coincides with the onset of widespread atmospheric testing of nuclear weapons (i.e. 1954). 1959-1960 and 1963-1964 are reported as periods of maximum fallout (Ritchie et al. 1973).

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APPENDIX B
MODEL PROGRAM LISTING

Newnan's Lake model - Simulation of the effect of spillway installation on in-lake sediment and phosphorus dynamics. Time scale is 40 years with spillway construction initiated at 15 years.

DEFDBL A-Z	Double precision

STRING VARIABLES

```
TITLE$ = "Effect of Spillway on Sediment and P-Dynamics"
SUBTITLE$ = "Newnan's Lake, FL"
T$ = "Years"
```

TIME PARAMETERS

T0 = 0	Start time (y)
THALF = 20	
TEND = 40	End time (y)
DT = .0005	Integration interval (y)
PLTINTVL = .05	Plotting interval (y)

INITIAL VALUES

PLOTCOUNT = 0	
PERMSED = 60000	Permanent sediments (g m^{-2})
SRPWATER = .006	Water column soluble reactive phosphorus conc. (g_{SRP} m^{-3})
PDETRITUS = .016	P-concentration of suspended dead organic matter (g_P m^{-3})
PPLANKTON = .048	P-concentration of suspended live organic matter (g_P m^{-3})
PINORG = .016	P-concentration of suspended inorganic matter (g_P m^{-3})
SRPPORE = .43	SRP in pore fluids in surfsed (g_{SRP} m^{-2})

PDETSED = 1.64	P-concentration in dead organic surfsed (g_P m^{-2})
PBENTHOS = .4	P-concentration in live organic surfsed (g_P m^{-2})
PINORGSED = 1.32	P-concentration in inorganic surfsed (g_P m^{-2})
VEGLITAREA = 20	Percent vegetated littoral area in lake
LITPART = 200	Seston + surfsed in littoral (g m^{-2})
DELLITPART = 5000	Seston + surfsed temp. trapped in littoral (g m^{-2})
AVGLEVEL = 20.24	Average lake stage (m msl)

CONSTANTS

AMPNODAM = .55	Amplitude of lake level fluctuation without dam (m)
AMPDAM = .39	Amplitude of lake level fluctuation with dam (m)
FREQLEVEL = 50	Frequency of water level fluctuation
RISEDAM = .13	Water-level rise following dam installation (m)
RAMPDURATION = .5	Ramp coefficient for water-level rise (y)
BWDNODAM = .5	Concentration of seston in outflow without dam (g m^{-3})
BWDDAM = .3	Concentration of seston in outflow with dam (g m^{-3})
HYDRESTMNODAM = .6	Hydraulic residence time without dam (y)
HYDRESTMDAM = .8	Hydraulic residence time with dam (y)
LAKEBED = 18.57	Initial elevation of sediment-water interface (m msl)
BULKDENS = 30000	Surfsed bulk density (g m^{-3})
BEDDENS = 500000	Permsed bulk density (g m^{-3})
SDR = .5	Specific rate of deposition (% y^{-1})
SVELP = 300	Settling velocity of plankton (m y^{-1})
SVELD = 2523	Settling velocity of detritus (m y^{-1})
SVELI = 2523	Settling velocity of inorganic particles(m y^{-1})
ALPHA = 36.85	Model coefficient (for Srr) (% y^{-1})
GAMMA = -.084	Model coefficient (for Srr)

MAXPAR = 16	Maximum photosynthetically active radiation (Par) at mean lake depth (cal $cm^{-2}hr^{-1}$)
OPTPAR = 8	Optimum Par at mean lake depth (cal $cm^{-2}hr^{-1}$)
BETA = .069	Extinction coefficient (% per g m^{-3} of seston)
MAXBENTHICBACT = 15.5	Max. P-uptake rate for benthic bacteria ($g_{SRP}\ g_{Pbenthicbact}^{-1}y^{-1}$)
MAXPLANKBACT = 15.5	Max. P-uptake rate for planktonic bacteria ($g_{SRP}\ g_{Pplankbact}^{-1}y^{-1}$)
MAXBENTHICAUTO = 3	Max. P-uptake rate for benthic algae ($g_{SRP}\ g_{Pbenthicauto}^{-1}\ y^{-1}$)
MAXPLANKAUTO = 10	Max. P-uptake rate for planktonic algae ($g_{SRP}\ g_{Pplankauto}^{-1}y^{-1}$)
KM1 = .31	Half-saturation constant for specific P-uptake rate by benthic bacteria ($g_{SRP}\ m^{-3}$)
KM2 = .036	Half-saturation constant for specific P-uptake rate by benthic algae ($g_{SRP}\ m^{-3}$)
KM3 = .31	Half-saturation constant for specific P-uptake rate by planktonic bacteria ($g_{SRP}\ m^{-3}$)
KM4 = .036	Half-saturation constant for P-effect ($g_{SRP}\ m^{-3}$)
BACTFRAC = .5	Bacterial fraction of plankton ($g_{Bact}\ g_{Plankton}^{-1}$)
AUTOFRAC = .42	Algal fraction of plankton ($g_{Algae}\ g_{Plankton}^{-1}$)
EUKFRAC = .08	Eukaryotic heterotroph fraction of plankton ($g_{Eukhet}\ g_{Plankton}^{-1}$)
PBACTCONT = .008	P-content of bacterio-plankton ($g_P\ g_{Bacteria}^{-1}$)
PAUTOCONT = .004	P-content of algae ($g_P\ g_{Algae}^{-1}$)
PEUKCONT = .004	P-content of eukaryotic heterotroph-plankton ($g_P\ g_{Eukhet}^{-1}$)
PDETRITUSCONT = .002	P-content of suspended detritus ($g_P\ g_{Detritus}^{-1}$)
PINORGCONT = .004	P-content of inorganic seston ($g_P\ g_{Inorgseston}^{-1}$)
BENTHICBACTFRAC = .75	Bacterial fraction of benthos ($g_{Bact}\ g_{Benthos}^{-1}$)

BENTHICAUTOFRAC = .2	Algal fraction of benthos (g_{Algae} $g_{Benthos}^{-1}$)
BENTHICEUKFRAC = .05	Eukaryotic heterotroph fraction of benthos (g_{Eukhet} $g_{Benthos}^{-1}$)
PBENTHICBACTCONT = .012	P-content of bacterial-benthos (g_P $g_{Bacteria}^{-1}$)
PBENTHICAUTOCONT = .004	P-content of benthic algae (g_P g_{Algae}^{-1})
PBENTHICEUKCONT = .004	P-content of eukaryotic heterotroph-benthos (g_P g_{Eukhet}^{-1})
PDETSEDCONT = .002	P-content of detrital surfsed (g_P g_{Detsed}^{-1})
PINORGSEDCONT = .003	P-content of inorganic surfsed (g_P $g_{Inorgsed}^{-1}$)
KF = .045	Model coefficient (for filter)
PLOAD = .05	Srp-contribution from runoff (g_{SRP} $m^{-2}y^{-1}$)
Q = 1350	Gain in Litpart from influx and production (g $m^{-2}y^{-1}$)
KT = .02	Model coefficient (for trapping of Litpart)
LITRESTM = .15	Litpart residence time (y)
M = .003	Slope coefficient for specific export rate
B2 = .007	Y-intercept for specific export rate (g $g^{-1}y^{-1}$)
SUBMLITAREA = 6000000	Submerged littoral area (m^2)
LAKEVOLUME = 40500000	Lake volume (m^3)
SLSR = .005	Specific littoral sedimentation rate (% y^{-1})
HALFLIFE = 5	Littoral area residence time (y)
B3 = 64.42	Model coefficient for gain in veglitarea
N = -5.07	Model coefficient for gain in veglitarea
PLITDETRITUSCONT = .002	P-content of detrital Litpart (g_P $g_{Litdetritus}^{-1}$)
PLITINORGCONT = .004	P-content of inorganic Litpart (g_P $g_{Litinorg}^{-1}$)
PLITPLANKTONCONT = .006	P-content of planktonic Litpart (g_P $g_{Litplankton}^{-1}$)

DETRITUSLITFRAC = .2	Detrital fraction of Litpart ($g_{Detritus}\ g_{Litpart}^{-1}$)
INORGLITFRAC = .2	Inorganic fraction of Litpart ($g_{Inorg}\ g_{Litpart}^{-1}$)
PLANKTONLITFRAC = .6	Planktonic fraction of Litpart ($g_{Plankton}\ g_{Litpart}^{-1}$)
PDELLITDETRITUSCONT = .002	P-content of detrital Dellitpart ($g_P\ g_{Dellitdetr}^{-1}$)
PDELLITINORGCONT = .004	P-content of inorganic Dellitpart ($g_P\ g_{Dellitinorg}^{-1}$)
PDELLITPLANKTONCONT = .006	P-content of planktonic Dellitpart ($g_P\ g_{Dellitplank}^{-1}$)
DETRITUSDELLITFRAC = .2	Detrital fraction of Dellitpart ($g_{Detr}\ g_{Dellitpart}^{-1}$)
INORGDELLITFRAC = .2	Inorganic fraction of Dellitpart ($g_{Inorg}\ g_{Dellitpart}^{-1}$)
PLANKTONDELLITFRAC = .6	Planktonic fraction of Dellitpart ($g_{Plank}\ g_{Dellitpart}^{-1}$)
V1 = 500	Specific reaction rate for Pinorg → Srpwater ($g_{Srpwater}\ g_{Pinorg}^{-1}y^{-1}$)
KR1 = 2.67	Equilibrium constant for Pinorg → Srpwater ($g_{Pinorg}\ g_{Srpwater}^{-1}$)
KR2 = 3.06	Equilibrium constant for Pinorgsed → Srppore ($g_{Pinorgsed}\ g_{Srppore}^{-1}$)
V2 = 500	Specific reaction rate for Pinorgsed → Srppore ($g_{Srppore}\ g_{PInorgsed}^{-1}y^{-1}$)
PLANKSPECREGRT = 500	Specific regeneration rate of Srp by euk.het.plankton ($g_{Srp}\ g_{Eukhet}^{-1}y^{-1}$)
BENTHOSSPECREGRT = 500	Specific regeneration rate of Srp by euk.het.benthos ($g_{Srp}\ g_{Eukhet}^{-1}y^{-1}$)
PLIFETM = .05	Mean lifetime of Pplankton (y)
BLIFETM = .025	Mean lifetime of Pbenthos (y)
KDETRITUS = 20	Specific rate of plankton decomposition ($g_{Pplankton}\ g_{Pdetritus}^{-1}y^{-1}$)
KDETSED = 20	Specific rate of benthos decomposition ($g_{Pbenthos}\ g_{Pdetsed}^{-1}y^{-1}$)

COMPUTED CONSTANTS

AMP = AMPNODAM

HYDRESTM = HYDRESTMNODAM

BWD = BWDNODAM

KIM = SUBMLITAREA / LAKEVOLUME
Conversion factor from m^2 littoral to m^3 profundal zone

PPLANKTONCONT = (PBACTCONT * BACTFRAC) + (PAUTOCONT * AUTOFRAC) + (PEUKCONT * EUKFRAC)
P-content of sestonic plankton ($g_P\ g_{Plankton}^{-1}$)

PAUTOFRAC = (PAUTOCONT * AUTOFRAC) / PPLANKTONCONT
Algal-P fraction of Pplankton

PBACTFRAC = (PBACTCONT * BACTFRAC) / PPLANKTONCONT
Bacterial-P fraction of Pplankton

PEUKFRAC = (PEUKCONT * EUKFRAC) / PPLANKTONCONT
Euk.Het.-P fraction of Pplankton

PBENTHOSCONT = (PBENTHICBACTCONT * BENTHICBACTFRAC) + (PBENTHICAUTOCONT * BENTHICAUTOFRAC) + (PBENTHICEUKCONT * BENTHICEUKFRAC)
P-content of benthos ($g_P\ g_{Benthos}^{-1}$)

PAUTOBENTHICFRAC = (PBENTHICAUTOCONT * BENTHICAUTOFRAC) / PBENTHOSCONT
Algal-P fraction of Pbenthos

PBACTBENTHICFRAC = (PBENTHICBACTCONT * BENTHICBACTFRAC) / PBENTHOSCONT
Bacterial-P fraction of Pbenthos

PEUKBENTHICFRAC = (PBENTHICEUKCONT * BENTHICEUKFRAC) / PBENTHOSCONT
Euk.Het.-P fraction of Pbenthos

A1 = 1 / (-OPTPAR^2) Model coefficient for Pareff

B1 = -A1 * MAXPAR Model coefficient for Pareff

GOSUB 5000 Call the plot-initialization subroutine

FOR T = T0 TO TEND STEP DT

AUXILIARY EQUATIONS

```
IF T >= 15 THEN

     AMP = AMPDAM
     HYDRESTM = HYDRESTMDAM
     BWD = BWDDAM
     RISE = RISEDAM

END IF

IF T >= 15 + RAMPDURATION THEN RISE = 0

LEVEL = AVGLEVEL + AMP * SIN((T / 7.957) * (FREQLEVEL))
```

Annual lake level fluctuation (m msl)

```
BOTTOM = (PERMSED / BEDDENS) + (SURFSED / BULKDENS) +
     LAKEBED
```

Mean elevation (m msl) of sediment-water interface

```
DEPTH = LEVEL - BOTTOM
```

Water column depth (m)

```
SRR = ALPHA * EXP(GAMMA * DEPTH)
```

Specific rate of resuspension by wind-generated waves and currents ($g\ g^{-1}y^{-1}$)

```
SURFSED = PINORGSED / PINORGSEDCONT + PBENTHOS /
     PBENTHOSCONT + PDETSED / PDETSEDCONT
```

Flocculent Surfsed subject to resuspension ($g\ m^{-2}$)

```
VSRPPORE = SRPPORE * BULKDENS / SURFSED
```

Srppore ($g_{Srp}\ m^{-3}$)

```
SPECBENTHICBACTPUR = (MAXBENTHICBACT * VSRPPORE) / (KM1 +
     VSRPPORE)
```

Spec. dark P-uptake rate for benthic bacteria ($g_{Srp}\ g_{Pbenthicbact}{}^{-1}y^{-1}$)

```
BENTHICBACTPUR = PBENTHOS * SPECBENTHICBACTPUR *
     PBACTBENTHICFRAC
```

P-uptake rate by benthic bacteria ($g_{Srp}\ m^{-2}y^{-1}$)

```
SPECBENTHICAUTOPUR = (MAXBENTHICAUTO * VSRPPORE) / (KM2 +
     VSRPPORE)
```

Spec. dark P-uptake rate for benthic algae ($g_{Srp}\ g_{Pbenthicauto}{}^{-1}y^{-1}$)

```
BENTHICAUTOPUR = PBENTHOS * SPECBENTHICAUTOPUR *
     PAUTOBENTHICFRAC
```

P-uptake rate by benthic algae ($g_{Srp}\ m^{-2}y^{-1}$)

SESTON = PINORG / PINORGCONT + PPLANKTON / PPLANKTONCONT + PDETRITUS / PDETRITUSCONT

Inorganic, dead, and live seston combined (g_{Seston} m^{-3})

PAR = MAXPAR * EXP(-BETA * SESTON)

Photosynthetically active radiation (cal $cm^{-2}hr^{-1}$)

PAREFF = A1 * (PAR^2) + B1 * PAR

Effect of Par on Specplankautopur

PEFF = SRPWATER / KM4 + SRPWATER

Effect of P on Specplankautopur ($0<Peff<1$)

SPECPLANKAUTOPUR = MAXPLANKAUTO * SQR(PAREFF * PEFF)

Spec. P-uptake rate for planktonic algae (g_{SRP} $g_{Pplankauto}{}^{-1}y^{-1}$)

PLANKAUTOPUR = PAUTOFRAC * PPLANKTON * SPECPLANKAUTOPUR

P-uptake rate by planktonic algae (g_{SRP} $m^{-3}y^{-1}$)

SPECPLANKBACTPUR = (MAXPLANKBACT * SRPWATER) / (KM3 + SRPWATER)

Spec. P-uptake rate for planktonic bacteria (g_{SRP} $g_{Pplankbact}{}^{-1}y^{-1}$)

PLANKBACTPUR = PBACTFRAC * PPLANKTON * SPECPLANKBACTPUR

P-uptake rate by planktonic bacteria (g_{SRP} $m^{-3}y^{-1}$)

FILTER = KF * VEGLITAREA — Percent P filtered from runoff by littoral

TRAPEFF = KT * VEGLITAREA — Percent of part. matter trapped by littoral

SER = M * SRR + B2 — Spec. export rate (g $g^{-1}y^{-1}$)

RATE-EQUATIONS

PPLANKTONFLUSH = PPLANKTON * BWD / HYDRESTM

Flushing of Pplankton through outlet ($g_{Pplankton}$ $m^{-3}y^{-1}$)

PINORGFLUSH = PINORG * BWD / HYDRESTM

Flushing of Pinorg through outlet (g_{Pinorg} $m^{-3}y^{-1}$)

PDETRITUSFLUSH = PDETRITUS * BWD / HYDRESTM

Flushing of Pdetritus through outlet ($g_{Pdetritus}$ $m^{-3}y^{-1}$)

SRPFLUSH = (1 / HYDRESTM) * SRPWATER

Flushing of Srpwater through outlet (g_{Srp} $m^{-3}y^{-1}$)

SEDRT = SDR * SURFSED
Sedimentation rate ($g_{Surfsed}$ $m^{-2}y^{-1}$)

PBENTHOSSEDRT = SDR * PBENTHOS
Pbenthos-sedimentation rate $g_{Pbenthos}$ $m^{-2}y^{-1}$)

PDETSEDRT = SDR * PDETSED
Pdetsed-sedimentation rate ($g_{Pdetsed}$ $m^{-2}y^{-1}$)

PINORGSEDSEDRT = SDR * PINORGSED
Pinorgsed-sedimentation rate ($g_{Pinorgsed}$ $m^{-2}y^{-1}$)

APBENTHOSRESUS = SRR * PBENTHOS
Areal rate of Pbenthos resuspension ($g_{Pbenthos}$ $m^{-2}y^{-1}$)

VPBENTHOSRESUS = SRR * PBENTHOS / DEPTH
Volumetric rate of Pbenthos resuspension ($g_{Pbenthos}$ $m^{-3}y^{-1}$)

APDETSEDRESUS = SRR * PDETSED
Areal rate of Pdetsed resuspension ($g_{Pdetsed}$ $m^{-2}y^{-1}$)

VPDETSEDRESUS = SRR * PDETSED / DEPTH
Volumetric rate of Pdetsed resuspension ($g_{Pdetsed}$ $m^{-3}y^{-1}$)

APINORGSEDRESUS = SRR * PINORGSED
Areal rate of Pinorgsed resuspension ($g_{Pinorgsed}$ $m^{-2}y^{-1}$)

VPINORGSEDRESUS = SRR * PINORGSED / DEPTH
Volumetric rate of Pinorgsed resuspension ($g_{Pinorgsed}$ $m^{-3}y^{-1}$)

AENTR = SRR * SRPPORE
Areal entrainment of Srppore by shear ($g_{Srppore}$ $m^{-2}y^{-1}$)

VENTR = SRR * SRPPORE / DEPTH
Vol.entrainment of Srppore by shear ($g_{Srppore}$ $m^{-3}y^{-1}$)

APPLANKTONSETTL = SVELP * PPLANKTON
Areal rate of settling of Pplankton ($g_{Pplankton}$ $m^{-2}y^{-1}$)

VPPLANKTONSETTL = SVELP * PPLANKTON / DEPTH
Volumetric rate of settling of Pplankton ($g_{Pplankton}$ $m^{-3}y^{-1}$)

APDETRITUSSETTL = SVELD * PDETRITUS
Areal rate of settling of Pdetritus ($g_{Pdetritus}$ $m^{-2}y^{-1}$)

VPDETRITUSSETTL = SVELD * PDETRITUS / DEPTH
Volumetric rate of settling of Pdetritus ($g_{Pdetritus}$ $m^{-3}y^{-1}$)

APINORGSETTL = SVELI * PINORG
Areal rate of settling of Pinorg (g_{Pinorg} $m^{-2}y^{-1}$)

VPINORGSETTL = SVELI * PINORG / DEPTH
Volumetric rate of settling of Pinorg (g_{Pinorg} $m^{-3}y^{-1}$)

PINORGSORP = V1 * (PINORG - KR1 * SRPWATER)
Adsorption/desorption rate between Pinorg and Srpwater ($g_{Srpwater}$ $m^{-3}y^{-1}$)

PINORGSEDSORP = V2 * (PINORGSED - KR2 * SRPPORE)
Adsorption/desorption rate between Pinorgsed and Srppore ($g_{Srppore}$ $m^{-2}y^{-1}$)

PREG = PEUKFRAC * PPLANKTON * PLANKSPECREGRT
Srp-regeneration rate by euk. het. plankton ($g_{Srpwater}$ $m^{-3}y^{-1}$)

PLANKPUR = PLANKBACTPUR + PLANKAUTOPUR
Phosporus uptake rate (g_{Srp} $m^{-3}y^{-1}$)

PPLANKTONDR = PPLANKTON / PLIFETM
Average death rate of Pplankton ($g_{Pplankton}$ $m^{-3}y^{-1}$)

PDETRITUSDECRT = PDETRITUS * KDETRITUS
Average detritus decomposition rate ($g_{Pdetritus}$ $m^{-3}y^{-1}$)

PPOREREG = PEUKBENTHICFRAC * PBENTHOS * BENTHOSSPECREGRT
Srp-regeneration rate by euk. het. benthos (g_{Srp} $m^{-2}y^{-1}$)

BENTHOSPUR = BENTHICBACTPUR + BENTHICAUTOPUR
Phosporus uptake rate (g_{Srp} $m^{-2}y^{-1}$)

PBENTHOSDR = PBENTHOS / BLIFETM
Average death rate of Pbenthos ($g_{Pbenthos}$ $m^{-2}y^{-1}$)

PDETSEDDECRT = PDETSED * KDETSED
Average detsed decomposition rate ($g_{Pdetsed}$ $m^{-2}y^{-1}$)

INFLOW = Q
Gain of Litpart from runoff and prod. ($g_{Litpart}$ $m^{-2}y^{-1}$)

TRAP = LITPART * TRAPEFF / LITRESTM
Accrual of Litpart ($g_{Litpart}$ $m^{-2}y^{-1}$)

THROUGH = (1 - TRAPEFF) * LITPART / LITRESTM

Particles not trapped by Litarea ($g_{Litpart}$ $m^{-2}y^{-1}$)

LITSEDRT = SLSR * DELLITPART

Littoral sedimentation rate ($g_{Dellitpart}$ $m^{-2}y^{-1}$)

DELTHROUGH = DELLITPART * SER

Delayed particles flow through Litarea ($g_{Dellitpart}$ $m^{-2}y^{-1}$)

PEX = PLOAD * (1 - FILTER) / DEPTH

Srp-export rate from littoral to pelagic (g_{Srp} $m^{-3}y^{-1}$)

GAIN = (100 / (1 + B3 * EXP(N * AMP)) / (HALFLIFE / .69))

Veglitarea gain with lake level fluctuation (%)

LOSS = VEGLITAREA / (HALFLIFE / .69)

Loss in Veglitarea with lake level fluctuation (%)

PPLANKTONIMPORT = (LITPART * KIM * (1 - TRAPEFF) / LITRESTM) * PLANKTONLITFRAC * PLITPLANKTONCONT

Import of Pplankton from Litarea ($g_{Pplankton}$ $m^{-3}y^{-1}$)

PDETRITUSIMPORT = (LITPART * KIM * (1 - TRAPEFF) / LITRESTM) * DETRITUSLITFRAC * PLITDETRITUSCONT

Import of Pdetritus from Litarea ($g_{Pdetritus}$ $m^{-3}y^{-1}$)

PINORGIMPORT = (LITPART * KIM * (1 - TRAPEFF) / LITRESTM) * INORGLITFRAC * PLITINORGCONT

Import of Pinorg from Litarea (g_{Pinorg} $m^{-3}y^{-1}$)

PPLANKTONDELIMPORT = KIM * DELLITPART * SER * PLANKTONDELLITFRAC * PDELLITPLANKTONCONT

Delayed import of Pplankton from Litarea ($g_{Pplankton}$ $m^{-3}y^{-1}$)

PDETRITUSDELIMPORT = KIM * DELLITPART * SER * DETRITUSDELLITFRAC * PDELLITDETRITUSCONT

Delayed import of Pdetritus from Litarea ($g_{Pdetritus}$ $m^{-3}y^{-1}$)

PINORGDELIMPORT = KIM * DELLITPART * SER * INORGDELLITFRAC * PDELLITINORGCONT

Delayed import of Pinorg from Litarea (g_{Pinorg} $m^{-3}y^{-1}$)

```
PVAR(1) = SESTON                 1st plot variable
PVAR(2) = SURFSED                2nd plot variable
PVAR(3) = SRPWATER               3rd plot variable
PVAR(4) = VEGLITAREA             4th plot variable
PVAR(5) = PPLANKTON              5th plot variable
PVAR(6) = SEDRT                  6th plot variable

IF T = 0 THEN GOSUB 5100

IF PLOTCOUNT >= PLTINTVL AND PLOTCOUNT < PLTINTVL + DT THEN

GOSUB 5100                       Call the plotting subroutine

PLOTCOUNT = PLOTCOUNT + DT

LOCATE 1, 1
     PRINT USING "####.##   "; T;

FOR I% = 1 TO NPVAR
     COLOR COLR(I%)
     PRINT USING "####.##   "; PVAR(I%);
NEXT I%

IF T > 15 AND T < 15.001 THEN
     LOCATE 30, 1
     PRINT USING "#####.###  "; T;
     FOR I% = 1 TO NPVAR
          COLOR COLR(I%)
          PRINT USING "#####.##  "; PVAR(I%);
     NEXT I%

END IF
```

ACCUMULATIONS

```
VEGLITAREA = VEGLITAREA + DT * (GAIN - LOSS)

LITPART = LITPART + DT * (INFLOW - TRAP - THROUGH)

DELLITPART = DELLITPART + DT * (TRAP - LITSEDRT -
     DELTHROUGH)

SRPWATER = SRPWATER + DT * (PEX + VENTR + PINORGSORP + PREG
     - PLANKPUR - SRPFLUSH)

PPLANKTON = PPLANKTON + DT * (PLANKPUR + VPBENTHOSRESUS +
     PDETRITUSDECRT + PPLANKTONIMPORT + PPLANKTONDELIMPORT -
     VPPLANKTONSETTL - PREG - PPLANKTONDR - PPLANKTONFLUSH)
```

```
PDETRITUS = PDETRITUS + DT * (PPLANKTONDR + VPDETSEDRESUS +
     PDETRITUSIMPORT + PDETRITUSDELIMPORT - PDETRITUSDECRT -
     VPDETRITUSSETTL - PDETRITUSFLUSH)

PINORG = PINORG + DT * (VPINORGSEDRESUS + PINORGIMPORT +
     PINORGDELIMPORT - VPINORGSETTL - PINORGSORP -
     PINORGFLUSH)

SRPPORE = SRPPORE + DT * (PINORGSEDSORP + PPOREREG -
     BENTHOSPUR - AENTR)

PBENTHOS = PBENTHOS + DT * (BENTHOSPUR + APPLANKTONSETTL +
     PDETSEDDECRT - PBENTHOSSEDRT - APBENTHOSRESUS -
     PPOREREG - PBENTHOSDR)

PDETSED = PDETSED + DT * (APDETRITUSSETTL + PBENTHOSDR -
     PDETSEDDECRT - PDETSEDRT - APDETSEDRESUS)

PINORGSED = PINORGSED + DT * (APINORGSETTL - PINORGSEDRT -
     APINORGSEDRESUS - PINORGSEDSORP)

PERMSED = PERMSED + DT * SEDRT

AVGLEVEL = AVGLEVEL + DT * (RISE / RAMPDURATION)


NEXT T


Data below:     a) number of plot variables (maximum of 6)
                b) for each plot variable:
                               1) a legend symbol
                               2) a color number
                               3) the minimum scale value
                               4) the maximum scale value

DATA 6
DATA "SEST",12,0,60
DATA "SURF",13,1000,4000
DATA "SRPW",10,0.0,0.2
DATA "LITA",15,0,20
DATA "PPLA",11,0,0.2
DATA "SEDR",14,0,1000

SLEEP

STOP
```

5000 Color plot initialization

```
CLS
SCREEN 12
COLOR 14
PRINT

PRINT SPC(40 - (LEN(TITLE$) / 2)); TITLE$
PRINT SPC(40 - (LEN(SUBTITLE$) / 2)); SUBTITLE$;
VIEW (138, 90)-(500, 405), , 14
WINDOW (0, 10)-(10, 0)

FOR I% = 1 TO 9
     LINE (I%, 0)-(I%, .1), 14
     LINE (I%, 10)-(I%, 9.9), 14
     LINE (0, I%)-(.1, I%), 14
     LINE (10, I%)-(9.9, I%), 14
NEXT I%

LOCATE 27, 17: PRINT TO;
LOCATE 27, 39: PRINT THALF;
LOCATE 27, 62: PRINT TEND;
LOCATE 29, 38: PRINT T$;

READ NPVAR

FOR I% = 1 TO NPVAR
     READ SYMBL$(I%), COLR(I%), MIN(I%), MAX(I%)
     COLOR COLR(I%)

IF I% > 3 THEN ZZ1 = 43 + 5.2 * I% ELSE ZZ1 = 19 - 5.5 * I%

     LOCATE 5, ZZ1: PRINT SYMBL$(I%);
     LOCATE 6, (ZZ1): PRINT MAX(I%);
     LOCATE 16, (ZZ1): PRINT INT((MAX(I%) + MIN(I%)) / 2)
     LOCATE 26, (ZZ1): PRINT MIN(I%);
NEXT I%

COLOR 10
LOCATE 16, 2.5: PRINT "0.1"
COLOR 11
LOCATE 16, 69.5: PRINT ".1"

RETURN
```

5100 Color plotting subroutine

```
PLOTCOUNT = 0

FOR I% = 1 TO NPVAR
     WINDOW (T0, MAX(I%))-(TEND, MIN(I%))
     PSET (T, PVAR(I%)), COLR(I%)
NEXT I%

IF T > 15 AND T < 15.001 THEN
     LOCATE 30, 1
     PRINT USING "#####.###  "; T;

FOR I% = 1 TO NPVAR
     COLOR COLR(I%)
     PRINT USING "#####.##  "; PVAR(I%);
NEXT I%

END IF

RETURN

END
```

BIOGRAPHICAL SKETCH

Johan F. Göttgens was born in Bloemendaal, The Netherlands, on June 13, 1953. He received a Bachelor of Science degree in biology/ecology from Utrecht State University in The Netherlands in 1976. Thesis research for his Master of Science degree in limnology was done at the University of Florida in Gainesville on an exchange fellowship. With the completion of this work and minors in environmental policy, didactics of biology, and research strategies in agriculture he received his Master of Science degree in Utrecht in 1981. After working as an Environmental Specialist for the Florida Department of Agriculture and Consumer Services for five years he returned to academics to continue his research interests in aquatic systems ecology and to pursue a career in teaching at the university level. He earned a Doctor of Philosophy degree from the University of Florida, Department of Environmental Engineering Sciences in 1992. He is married to Brigitte Syfrett and is the proud father of a daughter, Ida, and a son, Leo.

I certify that I have read this study and that in my opinion it conforms to acceptable standards of scholarly presentation and is fully adequate, in scope and quality, as a dissertation for the degree of Doctor of Philosophy.

Clay L. Montague, Chair
Associate Professor of Environmental Engineering Sciences

I certify that I have read this study and that in my opinion it conforms to acceptable standards of scholarly presentation and is fully adequate, in scope and quality, as a dissertation for the degree of Doctor of Philosophy.

Thomas L. Crisman, Cochair
Professor of Environmental Engineering Sciences

I certify that I have read this study and that in my opinion it conforms to acceptable standards of scholarly presentation and is fully adequate, in scope and quality, as a dissertation for the degree of Doctor of Philosophy.

Joseph J. Delfino
Professor of Environmental Engineering Sciences

I certify that I have read this study and that in my opinion it conforms to acceptable standards of scholarly presentation and is fully adequate, in scope and quality, as a dissertation for the degree of Doctor of Philosophy.

Frank G. Nordlie
Professor of Zoology

I certify that I have read this study and that in my opinion it conforms to acceptable standards of scholarly presentation and is fully adequate, in scope and quality, as a dissertation for the degree of Doctor of Philosophy.

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This dissertation was submitted to the Graduate Faculty of the College of Engineering and to the Graduate School and was accepted as partial fulfillment of the requirements for the degree of Doctor of Philosophy.

August 1992

for Winfred M. Phillips
Dean, College of Engineering

Madelyn M. Lockhart
Dean, Graduate School